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Ecosystem Recovery Planning for Listed Salmon: An Integrated Assessment Approach for Salmon Habitat

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CHAPTER 1. INTRODUCTION

One of the main purposes of the Endangered Species Act (ESA) of 1973 is "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved." The ESA consequently requires the development and implementation of recovery plans in order to affect the conservation of listed species, and details that recovery plans must include:

1) a description of such site-specific actions as may be necessary to achieve the plan's goal for conservation and survival of the species;
2) objective, measurable criteria which, when met, would result in a determination that the species be removed from the list; and
3) estimates of the time and cost required to carry out those measures needed to achieve the plan's goal and to take the intermediate steps toward that goal.

For ESA-listed salmon in the western United States, this requirement is no small task, as salmon habitat is ubiquitous and the actions required to protect or restore the ecosystems on which salmon depend are in conflict with nearly every land use in the region.

The ESA provides little guidance concerning the content of recovery plans for individual species. Therefore, the National Marine Fisheries Service (NMFS) provides additional scientific guidance on setting population recovery goals (McElhany et al. 2000), based on the concept of viable salmonid populations (VSPs). McElhany et al. (2000) identify four types of recovery goals that must be met in each evolutionarily significant unit (ESU) of listed salmon: abundance, productivity, spatial structure, and diversity. An ESU, equivalent to a "distinct population segment" under the ESA, is "a population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species" (Myers et al. 1998).

In addition to the VSP guidance, NMFS provides guidance to the Technical Recovery Teams (TRTs), which are tasked with developing the technical aspects of a recovery plan for each ESU of listed salmon (NMFS 2000, referred to as the TRT Guidance Document). In this guidance, the habitat elements of the TRT work program are identified as:

1) describe fish-habitat productivity relationships,
2) identify factors for decline and limiting factors, and
3) identify early actions for recovery.

The TRT Guidance Document goes on to indicate that characterizing the habitat/fish productivity relationship includes assessing the spatial distribution of fish abundance for each population in the ESU, associating fish abundance with habitat characteristics, and identifying human factors that have the greatest impact on key freshwater and marine habitat. However, it does not specify appropriate spatial scales or resolution levels of data analyses. Moreover, it does not clearly elucidate the questions that such analyses are intended to answer, especially as they relate the population goals for diversity and spatial structure. The TRT Guidance Document also stops short of specific questions for identifying limiting factors and identifying habitat recovery actions.

Here we describe how to identify and prioritize the habitat elements of a recovery plan for salmon listed under the ESA. We begin with a conceptual framework for understanding relationships among land uses, watershed functions, habitat conditions, and biota (Chapter 2). The conceptual framework relies on principles of watershed and ecosystem management, and organizes the habitat-related questions that each recovery plan should attempt to answer. These questions first address disruptions to ecosystem functions and biological integrity. Understanding these changes provides the basis for identifying restoration actions that are necessary to recover the ecosystem upon which salmon depend. A second group of questions addresses how habitat changes might have affected each of the four population recovery goals for an ESU of salmon: abundance, productivity, spatial structure, and diversity. Some of these questions are relevant at the spatial scale of the entire ESU, whereas other questions must be answered for each population within an ESU. Answers to these questions are useful in prioritizing restoration actions identified from assessments of ecosystem functions and biological integrity.

After listing the important questions, we describe methodologies that are appropriate for answering each question. These methods cover initial assessments designed to rapidly describe conditions across entire ESUs (Chapter 3), as well as inventories that are required to identify and prioritize restoration actions (Chapters 4). We then describe how to use the information gathered in these assessments to identify and prioritize restoration actions that are necessary for ecosystem recovery and for recovery of salmon populations (Chapter 5), and how to use new information to update the recovery plan (Chapter 6). Finally we discuss the issues of scale and uncertainty, and how to account for them in a habitat recovery plan (Chapters 7 and 8). Several case studies are also included in Appendices to give concrete examples of how different questions can be answered using the methodologies we describe in this document.

CHAPTER 2. AN INTEGRATED ASSESSMENT APPROACH FOR HABITAT RECOVERY PLANNING

Timothy J. Beechie

In this chapter we briefly describe an approach to understanding ecosystem functions and habitat change, and explain a series of questions that should be answered in the process of developing a habitat recovery plan. We begin by describing an assessment approach that focuses on understanding how aquatic ecosystems that support salmon have been degraded. These assessments identify the types of recovery actions that are necessary for ecosystem recovery. Assessing how habitat changes have affected any single species is secondary, but necessary for understanding which recovery actions are likely to have the greatest benefits for listed species. We also briefly describe the scientific and practical reasons for choosing this approach. After describing the general approach, we describe a conceptual framework for understanding relationships among ecosystem processes, land uses, habitat conditions, and biota. This conceptual framework organizes the series of questions that must be answered, and clarifies the purpose of each assessment method. We also describe the relationships among the different assessments and clarify the sequence in which the assessments can be conducted.

Restoring Ecosystems to Support Recovery of Listed Salmon

Over the past decade, a number of scientists have pointed out that the listing of salmon is largely a result of trying to manage individual species and habitat characteristics rather than managing whole ecosystems (Doppelt et al. 1993). However, emerging management concepts such as *ecosystem management* or *managing for biodiversity* remain ill-defined by their respective scientific communities, and have not been developed into systems that are useful to managers. In this situation, local managers feel compelled to act, but they must continue to apply short-term restoration measures (e.g., placement of log structures, construction of spawning riffles, bank protection) that fail to recognize the integrated nature of physical and ecological processes in watersheds, and the importance of life-history variability to salmon runs (Frissel and Nawa 1992, Lichatowich et al. 1995). Therefore, managers continue to define habitat "problems" based on perceptions of what habitat for a particular species should look like, rather than on the causes of changes to habitat (Frissel and Nawa 1992), or on habitat changes that have the largest effect on salmon populations (Reeves et al. 1991).

Restoration that carefully considers the watershed or ecosystem context is more likely to be successful at restoring individual or multiple species and preventing the demise of others (Nehlsen et al. 1991, Doppelt et al. 1993, Lichatowich et al. 1995, Reeves et al. 1995). We therefore assert that the most important analyses conducted in support of habitat recovery planning are those that address disruptions to ecosystem functions and biological integrity. The goal of these assessments is to identify alterations of key processes that affect stream ecosystems, and specify the management actions required to restore those processes that sustain aquatic habitats and support biological integrity. In this approach, restoring specific salmon populations (or any other single organism) is subordinate to the goal of restoring the aquatic ecosystem that supports multiple salmon species. However, information on habitat changes or conditions that limit specific salmon populations can be useful for identifying actions that may have the greatest effect on salmon recovery (e.g., Reeves et al. 1991). As long as all restoration actions are consistent with the overriding goal of restoring ecosystem processes and functions, habitats will be restored for multiple species, but in a sequence that favors one over the others.

Scientific Basis

The scientific basis for this approach can be summarized in two important characteristics of salmon and their habitats (Beechie et al. 1996):

1. Salmonid stocks are adapted to local environmental conditions, including the dynamic nature of their environment (Miller and Brannon 1982, Healey 1991, Reeves et al. 1995).
2. Spatial and temporal variations in landscape processes create a dynamic mosaic of habitat conditions in a river network (e.g., Naiman et al. 1992).

These statements imply that salmonid species or stocks are adapted to spatially and temporally variable habitats (Beechie et al. 1996), and may further imply that such environmental variability is important to the long-term survival of stocks or races (Reeves et al. 1995). Perhaps most importantly, different salmon populations (even some located very close to each other) are adapted to the different spatial and temporal *sequences* of habitat conditions found in each watershed within an ESU.

These two points encapsulate the basic scientific disagreement with traditional management approaches that attempt to manipulate specific elements of salmon habitat. This fundamental disagreement has been variously described in the literature as problems with "one-size-fits-all" habitat standards (Bisson et al. 1997), not managing for spatial or temporal variation in habitats (Reeves et al. 1995, Bisson et al. 1997), and addressing symptoms of a disrupted ecosystem rather than the causes (Frissell and Nawa 1992). At its core, this disagreement stems from the fact that traditional approaches manage for spatially uniform and temporally static conditions. However, traditional approaches do not consider that local populations are adapted to the natural potential habitat conditions within their range, and that those conditions vary in space and time (Beechie et al. 1996). By contrast, identifying the root causes of degradation (i.e., disruptions to ecosystem processes and functions) and focusing restoration on those processes that form and sustain habitats allows each part of the river network to express its natural potential habitat, and will help conserve and restore the natural spatial and temporal variation of habitats to which salmon are adapted.

We stress that identifying the root causes of ecosystem degradation is important for two main reasons. First, we do not understand most of the linkages between landscapes, habitat, and salmon populations with any great certainty, and we cannot predict exactly how land uses alter habitat conditions or how those habitat changes alter salmon populations. In fact, it can be argued that we are not yet even aware of all the aspects of aquatic ecosystems that significantly affect salmon populations. This lack of knowledge has in the past led to significant habitat degradation. For example, in the 1970s the role of wood debris in habitat formation was poorly understood, and biologists generally perceived that wood accumulations in rivers hindered adult upriver migrations. Therefore, biologists recommended widespread wood removal to help salmon (even as recently as the 1980s), but inadvertently severely degraded the habitats they were trying to improve. Had we rather chosen to recognize that wood debris was a natural part of the ecosystem to which salmon are adapted, we could have avoided significant habitat loss by choosing management actions that preserved riparian forest processes and natural wood functions in channels.

Second, traditional restoration actions such as bank protection or spawning gravel placement attempt to build habitats that do not move in space or time, whereas natural habitats are often created by movement of river channels, wood debris, and sediment. Therefore, many restoration actions fail to restore habitats because they do recognize the integrated nature of physical and ecological processes in watersheds (Frissell and Nawa 1992, Beechie et al. 1996). This lack of knowledge leads to two main types of failure: 1) site-prescribed engineering solutions can be overwhelmed by altered watershed processes that are far removed from degraded habitats (e.g., increased sediment supply from upslope sources can bury engineered structures and pools), or 2) such measures can prevent habitat formation that would otherwise naturally occur (e.g., bank protection prevents formation of new off-channel habitats). Avoiding these types of project failure requires that we focus on restoring ecosystem processes and functions that form and sustain salmonid habitats, rather than on the habitats themselves.

Practical Considerations

Over the past two decades, scientists have proposed several new management approaches that seek to alleviate failures associated with managing individual habitats and species. In general, the literature has transitioned from watershed management (e.g., Swanson 1981), to ecosystem management (e.g., Johnson et al. 1985), to managing for biodiversity (e.g., McNeely et al. 1990). However, management regimes for salmon and their habitats lag far behind the literature, and it can be argued that most jurisdictions are just now beginning to understand and utilize the advantages of watershed management. During that same time, understanding and managing for biological integrity of running waters became an accepted approach for diagnosing and monitoring problems related to the Clean Water Act (CWA) (Karr 1991). Currently, the region is faced with widespread listings under the ESA, which states as one of its purposes to "conserve the ecosystems" upon which endangered species depend.

On the surface, this array of management approaches and legal requirements presents a management dilemma: how do we choose an approach to implement, and will it adequately respond to the ESA and CWA? However, the three management approaches are in reality closely related, and the purposes of the ESA and CWA are also similar in some respects. With respect to the management approaches, watershed processes and biodiversity are both essential components of the aquatic ecosystems that support salmon. That is, managing an entire riverine ecosystem requires that watershed processes be restored and maintained, and that the biodiversity of salmon and other organisms be conserved. With respect to legislation, an ecosystem approach will accomplish the habitat-related purposes of both acts, which are to conserve the ecosystems upon which listed species depend (ESA), and to restore and maintain the physical, chemical, and biological integrity of the nation's waters (CWA).

Conceptual Framework for Habitat Recovery Planning

Developing the habitat elements of a salmon recovery plan requires considerable information about how different land uses have altered landscape and ecological processes that form and sustain the habitats upon which salmon depend. In this chapter we describe two basic groups of questions that must be answered to develop a habitat recovery plan (Table 2-1), and explain how each describes specific components of watershed function. The first group of questions focuses on identifying disruptions to ecosystem function and the types of habitat restoration that are necessary for ecosystem recovery. These questions motivate assessments that identify where the biological integrity of ecosystems has been degraded and where specific ecosystem processes or functions are disrupted. The second set of questions concentrates on how habitats have changed since pre-settlement times and how those habitat changes have affected salmon and other biota. These questions motivate historical reconstructions of habitat types and abundance, as well as assessments of relationships between habitat and salmon population characteristics.

Key Assessments for Habitat Recovery Planning

For organizational purposes it is useful to diagram the relationships between the various assessments that can be used to inform recovery planning. The first set of questions (those regarding ecosystem functions and biological integrity) can be separated into two components: 1) diagnosing aquatic ecosystem impairment through a multi-metric biological indicator such as B-IBI (Karr xxxx), and 2) diagnosing causes of ecosystem impairment through specific assessments of watershed and ecosystem function (Beechie and Bolton 1999) (Figure 2-1). Assessing the biological integrity of aquatic ecosystems provides two important pieces of information required in a recovery plan. First it identifies where the aquatic ecosystem has been impaired, and second it can suggest which types of ecosystem functions may be impaired. A multi-metric index such as B-IBI assesses the biota directly, and results of these assessments can be correlated with landscape and land use factors to indicate potential causes of impaired biological integrity. Assessing ecosystem processes that form salmon habitats directly identifies causes of degradation and restoration actions that are required to recover ecosystem functions and biological integrity.

The second group of questions (those regarding changes in habitat and salmon populations) also fall into two groups: 1) assessments that quantify habitat change and then use habitat-based models to estimate changes in fish populations (e.g., limiting factors analysis, life-cycle models, EDT), and 2) correlation analyses that relate landscape and land use characteristics to fish population performance without directly quantifying changes to habitat (e.g., SWAM). It should be noted that neither of these assessments directly identifies causes of habitat degradation or specific restoration actions. However, these assessments have several important uses in recovery planning. First, they provide habitat-based estimates of potential population size for comparison to estimates from population viability analyses (see McElhany et al. 2000 for background on use of population viability analyses in describing VSPs). Second, they indicate which habitat changes are most likely responsible for declines in salmon populations, and therefore which categories of restoration actions are most likely to result in increased salmon populations. Finally, the ESU-wide correlation analyses can be used to help identify which populations are most constrained by habitat loss and therefore may be most difficult to recover.

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Table 2-1. Primary questions to answer in developing habitat recovery plans.

Question	Analysis area	Data types
Assessing disruptions to ecosystem functions and biological integrity		
Where has biological integrity been degraded?	Watershed	Field
Where have watershed processes and ecosystem functions been impaired?	Watershed	Field/remote sensing
Assessing changes in habitat availability and potential impacts on population characteristics		
How might habitat changes have altered the abundance of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the productivity of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the diversity of life history patterns?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the spatial structure of populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
What scenarios of habitat characteristics would support a viable ESU (viable meaning adequate levels of all 4 VSP parameters)?	ESU	Mainly remote sensing

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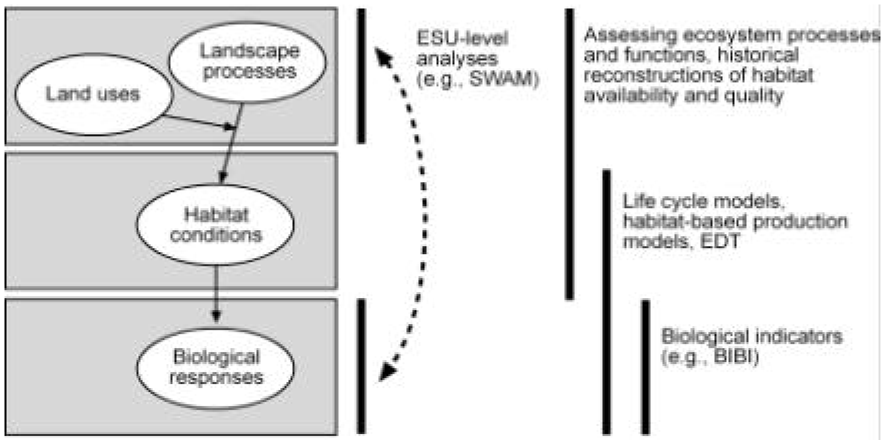


Figure 2-1. Schematic diagram of linkages among landscape processes, land use, habitat change, and biological responses. Assessing the biological response directly (e.g., using a biological indicator) identifies where ecosystem functions have been impaired, and may suggest causes of impairment. Assessments of habitat loss and resultant salmon population declines can be conducted by relating historical and current habitat abundance and condition to salmon utilization and survival. Assessing disrupted ecosystem functions and processes identifies causes of habitat change that result in diminished biological integrity and declines in salmon populations. For ESU-wide analyses of land use effects on salmon populations, landscape and land use factors can be correlated with indicators of population performance (e.g, SWAM-like analyses).

Sequencing the Assessments

Figure 2-1 illustrates that natural landscape processes are linked to biological responses by their effects on aquatic habitats, and that land uses alter habitat conditions by disrupting natural processes. That is, aquatic habitats are sensitive to landscape process inputs (e.g., water, sediment, wood, nutrients), and different types of aquatic habitats have differing capacities to support salmon and other biota. Assessing changes in habitat-forming processes (e.g., hydrology, sediment supply, or wood recruitment) indicates how land uses have altered habitats, and estimating historical and current habitat availability indicates where habitat changes have been greatest. An index of biotic integrity and can also indicate the degree to which different practices have altered the aquatic ecosystem, and can identify ecosystem disruptions that may be missed in the other two assessments. Once the important habitat-altering processes have been identified, inventories of specific land use actions that cause these changes can be used to identify habitat recovery options, which may include protection of good quality habitats as well as restoration of degraded habitats. Assessments of changes in the abundance or quality of each habitat type can subsequently be used to make simple estimates of changes in fish abundance, provided that relationships among different habitat types and salmon abundance can be estimated. These estimates can then be used to help prioritize recovery actions identified in the inventories.

In practice, sequencing of the assessments is determined by management needs and the rate at which necessary information can be acquired. At present, habitat restoration is underway, and the management need for reliable identification of important habitat recovery actions is immediate. However, identification of site-specific restoration actions requires time-consuming field inventories that are far from complete in many cases. Given this immediate need and the relatively long time frame required to complete the many required inventories, the first assessments should be rapid assessments of land uses, habitat change, and biological responses across entire ESUs (Table 2-2). These assessments (discussed in detail in Chapter 3) often rely on remote sensing data such as satellite imagery or digital elevation models as primary data sources (e.g., Lunetta et al. 1997). Other coarse resolution geospatial data sets (such as maps of land use types, mean annual temperature, or mean annual precipitation) are also frequently used. Site-specific data such as barrier inventories or stream habitat data may also be available in some areas, but in most cases these data exist for relatively few sites and must be extrapolated to most of the ESU. Therefore, ESU-level questions typically will be addressed with broad correlative studies that relate landscape and land use attributes to fish population parameters, without site-specific understanding of habitat changes that have occurred in streams. Referring to Figure 2-1, these assessments will relate landscape attributes and land use practices directly to fish abundance or survival, and ignore causal mechanisms that link landscape processes and land uses to habitat change or habitat change to fish population response. These ESU-wide assessments can indicate general patterns of population declines resulting from different land uses, but the data are generally too coarse to allow detailed analyses of habitat change and its effects on fish populations (e.g., Lunetta et al. 1997, Pess et al. 1999a).

Watershed-level assessments are more detailed and time-consuming, and managers should expect such that such inventories and assessments will take several years to complete. Such efforts should focus on the full range of potential habitats in the planning area, including freshwater, estuary, and nearshore marine areas. The conceptual model itself does not limit the spatial or temporal resolution of the results. Rather, specificity of results is driven by the resolution of the data, which in turn is driven by the assessment goals. For example, coarse resolution remote sensing data might be used to investigate patterns in broad patterns of riparian alteration within a watershed, which can be used to estimate the overall magnitude of the problem and to estimate restoration costs for a watershed (e.g., the riparian assessments in Appendix A). However, these data and assessments do not have sufficient detail to pinpoint the exact nature of the problem at any single site, and field inventories must be conducted in order to identify site-specific recovery actions. Similarly, other types of restoration actions such as removing migration barriers or landslide hazard reduction require field inventories to identify specific actions. (Note also that this remains true even if EDT analyses have been completed.) As a general rule, larger scale assessments help managers understand where landscape processes have been most altered, and site-specific inventories identify specific actions that are needed to restore those processes.

Implementing Recovery Actions

We support habitat recovery strategies that have an overarching goal consistent with the ESA (i.e., conserve the ecosystems upon which listed depend), as well as the CWA (i.e., protect the biological integrity of aquatic systems). Consistency with the purposes of these two acts will simplify the assessments needed to identify necessary habitat protection and restoration actions, and help avoid conflicts that arise from managing for the specific (and often conflicting) habitat requirements of multiple species. Examples of such a goal are to “protect and restore the processes that form and sustain habitats to which salmonid stocks are adapted” (Skagit Watershed Council 2001), “restoration and protection of habitat conditions and processes upon which the fish depend” (Lower Columbia Fish Recovery Board 2001), or “to have a diversity of habitats and natural processes necessary to sustain healthy populations of native species" (Willamette Restoration Strategy 2001).

Strategies should also have clearly stated near-term and long-term objectives. Near-term objectives typically should include taking those actions that we already know are consistent with conservation of ecosystems that support salmon, such as inventory and removal of migration blockages, habitat protection through easements or purchase, and protection and restoration of riparian forests and their functions (see also Chapter 5 for more detail). Longer term objectives may include targeted assessments or management experiments to clarify which actions are most beneficial to aquatic ecosystems, as well as implementing larger restoration projects that require changes in infrastructure or land uses, such as modifying levee systems to reopen access to estuary habitats and tidal channels.

Once the goals and objectives of the strategy are defined, it is important to identify specific questions that must be answered in order to proceed with protecting and restoring aquatic ecosystems. These questions will generally be more targeted and specific than those listed in Table 2-1. For example, several of the questions listed in Table 2-1 require that we know how much habitat has been blocked to salmon access. An obvious set of questions can be written to address this topic, including 1) where are the blockages to salmon migration, 2) how much habitat is above each blockage, and 3) how much will it cost to repair each blockage? These questions drive the need for a comprehensive and systematic inventory of stream crossing structures (including small dams, levees, tide gates, culverts, etc.), which can be conducted using widely accepted standard methods such as the barrier inventory methodology of Washington Department of Fish and Wildlife (WDFW 1998). Similar sets of questions can be written for other aspects of ecosystems such as riparian functions or changes in supply of sediment, as well as for understanding the condition of aquatic habitats through inventories of habitat and biological indicators such as invertebrate or fish communities.

We recognize that many jurisdictions and restoration groups are overwhelmed by the many assessments and inventories needed to enact a restoration strategy. However, the task is not so daunting as it first appears, provided that one recognizes the long-term nature of it. In essence, the steps are:

1. Develop the restoration strategy.
2. Identify what you need to know to implement the strategy.
3. Identify what you already know.
4. Act on what you know while you conduct inventories that fill in the data gaps.
5. Revise the plan and actions as new information comes in.

Steps 1 and 2 describe the overall restoration strategy and the types of information required to implement it. In some respects this is the most difficult step. In this Chapter we explained a restoration strategy that focuses on restoring the ecosystems upon which salmon depend, and briefly described the questions that must be answered in order to restore them. This general approach is supported by the scientific literature, and alleviates several management conflicts that may arise from attempts to separately manage individual species and habitat characteristics. Adopting a conceptual framework similar to that described here will help TRTs and local jurisdictions to more rapidly move through Steps 1 and 2, organize their information needs, and ultimately to better understand the habitat changes that have contributed to salmon declines in the region.

In the following chapters we provide additional guidance on the specific analyses required to address certain questions that will arise in recovery planning (i.e., greater detail on

Step 2 above). In Chapter 3 we describe how ESU-level assessments can be used to create a broad understanding of habitat issues affecting salmon populations across an ESU. These types of assessments will typically be under the purview of TRTs, and only provide a general sense of the types and magnitudes of land use alterations affecting the performance of different salmon populations. In Chapter 4 we describe more detailed assessments to be conducted within individual watersheds that identify causes of habitat loss or degradation, as well as changes in habitat or biological conditions. Data that identify causes of habitat change can be used to develop restoration and protection actions, whereas data describing habitat condition or biological status of streams can be used to prioritize recovery actions and monitor progress toward recovery.

Steps 3 and 4 are implementation steps. In Chapter 5 we discuss interim actions that are known to be consistent with protecting and restoring the ecosystems upon which salmon depend, and also describe how to use the information collected in various assessments to identify and prioritize appropriate restoration actions (Step 3 and 4). Step 3 requires examining all the questions identified from the strategy and listing which questions are already answered or partly answered. In step 4 local jurisdictions can identify areas where causes of habitat degradation or loss are already known, and therefore where habitat restoration or protection actions can be implemented immediately. Over the long term, identification of habitat protection and restoration actions relies on systematic inventories of causes of biological conditions in streams and causes of habitat loss. These inventories must be comprehensive and systematic in order to reliably identify the suite of possible restoration actions and to evaluate their relative importance to ecosystem recovery. (see also Chapter 4 and Appendix A).

Finally, new information acquired over the next several years will continually modify our understanding of how ecosystems have been altered, and of how those alterations have affected salmon populations. We can be certain that much of what we learn will change our prioritization of recovery actions, and that our recovery plans will need to be updated as new information comes in. In Chapter 6 we consider how new information can be incorporated into recovery plans and prioritized lists of habitat recovery actions.

CHAPTER 3. ASSESSING HABITAT (ESU-LEVEL)

Beth Sanderson and E. Ashley Steel

There are a number of motives for conducting broad-scale analyses. From a biological point of view, the recovery of salmonids must occur at the ESU scale. For this to happen, we need to answer questions, formulate hypotheses, and develop recovery scenarios for the entire ESU. At the moment, fine-resolution data are not comprehensively available across the large geographic areas these analyses must cover. Recovery planning efforts need answers before new field data can be collected. Broad-scale analyses can use currently available data to quickly address questions that span large geographic areas in a relatively short period of time. Standardized and consistent approaches are critical for comparisons both within and across ESUs. Coarse-scale analyses also form a starting point for further, more detailed work by organizing existing data, identifying broad patterns and generating new hypotheses.

Broad-scale analyses are intended to provide initial answers to the primary recovery questions identified in Chapter 2 (see Table 2-1). How might habitat changes have altered the abundance, productivity, spatial structure and diversity of individual populations? And, what scenarios of habitat characteristics would support a viable ESU? The approach is designed to provide information that links landscape characteristics (land use and land form), habitat, and fish (see Figures 2-1 and 3-1). The specifics for individual ESUs will vary according to the relevant questions, major land forms and land uses, and data availability.

This chapter describes how linkages between landscape, habitat and fish can be estimated from available data. The first section describes an approach to correlating fish abundance with habitat condition over broad spatial scales. The second section suggests methods for organizing available data to provide estimates of current habitat quantities. This section also provides example methodology for evaluating how different land uses may have changed the availability and distribution of aquatic habitats. Then in the final section, we describe available methods for quantifying the abilities of these habitats to support healthy and productive salmon populations.

Correlation Between Fish Populations and Habitat

Examining available data on fish populations and habitat conditions is a first step toward understanding the quantitative relationships between salmonid populations and the physical, chemical, and biological components of their habitat. Many potential metrics of population performance exist: genetic diversity, juvenile abundance, adult abundance, productivity, juvenile productivity per adult spawner, etc. Similarly, there are numerous habitat metrics: percent pools, watershed condition, water temperature, number or concentration of contaminants, etc. Correlation between population performance and habitat condition cannot identify cause and effect relationships because of correlation among habitat descriptors, correlation between landform and land use, and the potential for unmeasured variables to explain existing patterns. Correlative analyses can be used to make predictions about where habitat conditions might limit or enhance salmon populations, to generate hypotheses for further testing, and to suggest important factors to control when setting up small-scale experiments, monitoring projects, or large management experiments (Figure 3-1b).

One example of this type of analysis is the Salmonid Watershed Analysis Model (SWAM) (Feist et al. in review). SWAM is a series of spatial and statistical analyses that relate salmonid population performance metrics in a particular basin to landscape and land use characteristics derived from existing geospatial data layers. This analysis identifies descriptors of landform and land use that are correlated with fish population performance in a given watershed.

SWAM has been used in the Salmon River basin in Idaho, the Snohomish River basin in Washington, and the Willamette River basin in Oregon. In these three basins, SWAM used indices of adult fish abundance (redd counts and adult fish counts at index sites) as the metric of fish population performance, and multiple descriptors of landscape conditions across the entire watershed draining to the index reach as a surrogate for habitat condition. An alternate habitat metric, conditions in the riparian area directly associated with the index reach, was also tested in the Salmon and Snohomish basins.

The spatial and statistical analyses involved in the SWAM approach are comprised of six steps. First, conceptual mechanistic relationships between landscape features and population abundance during all freshwater life-history stages are identified from the literature and from local habitat biologists. These conceptual relationships define the habitat data layers to be used as potential predictor variables. Second, spatial heterogeneity in the salmonid population data is examined to determine if certain areas in the basin consistently exhibit better population performance than other areas. Third, the landscape is characterized over the relevant area for each index reach. Fourth, landscape data layers are overlaid with the geo-referenced fish abundance (e.g., redd counts) data. Fifth, a statistical model is used to describe annually consistent relationships between landscape characteristics and fish abundance. And finally, these relationships are rolled into a predictive model and applied to the entire basin of interest (Figure 3-1b). Many variations on the population and landscape metrics used in this type of analysis are possible. The best choice for a particular basin will be driven by available data.

SWAM or SWAM-like analyses provides a broad-brush estimation of fish occupancy within a basin and a first-cut estimate of the coarse-scale factors affecting abundance. In many cases, correlative models relating fish population performance to habitat conditions may also help identify the best remaining reaches or subwatersheds in a particular basin. If clear relationships between fish populations and habitat conditions exist, these analyses may suggest indicator habitat features and may identify areas that were historically productive. Ecological insights developed from these analyses may suggest likely habitat factors limiting population performance in a particular basin.Experience from these studies can be used to identify habitat characteristics to control for when setting up experiments and M&E programs. Predictions of areas likely to support strong populations can suggest areas where detailed watershed assessments and habitat inventories should be conducted (See Watershed Assessment Chapter in the document).

ESU-Level Habitat Inventory

An ESU-level habitat inventory is a method for assembling and organizing available habitat data in a geographic information system. Potential data sources include remotely sensed data layers and field-derived data. The most important data layer is the routed stream coverage. A comprehensive barrier inventory that identifies the type of barrier (natural, culvert, dam, etc.), location, and extent to which fish passage is blocked is needed to evaluate and compare the amount of habitat currently and historically accessible to salmonids. Data layers describing land use and landform can be used to classify stream segments and to model instream habitat characteristics. Habitat field surveys will be necessary to calibrate models for predicting instream habitat characteristics in areas where there is no data. Hydromodification inventories (i.e., levees, straightening, riprap, etc.), where available, will be useful for estimating changes in quantities of different habitat types.

Routed Stream Network

For most ESUs, it will be necessary to work at the 1:24,000 scale. This resolution is required to make sure that the stream inventory includes all areas that are used by the species at any of the life stages. In many areas, routed hydrographic data have already been generated by state or federal agencies at the 1:24,000 scale. For regions where such data are not available, 10-meter digital elevation models from the United States Geological Survey (USGS) can be used to generate stream networks, as well as additional stream variables that are useful, including stream gradient, stream order, valley width, drainage area, and side slope gradient. Once all streams are identified, individual stream reaches can be coarsely classified according to habitat type. Field-derived data and digital ortho-photographs can provide information to validate the stream network and aid in classification of stream networks. Examples of coarse-scale habitat types include large mainstem habitats, small mainstem habitats, tributaries, and off-channel habitats. One example of a hierarchical, nested framework for classifying streams is presented in Chapter 4. This classification framework is being used in recovery analyses, although regional differences in habitats and available data may necessitate modification of the classification categories.

Once all reaches have been classified according to habitat type, the total amount of each habitat type can be quantified within individual watersheds or regions and across the whole ESU. In addition, the distribution of habitat types can be compared among different regions or watersheds within an ESU area. Such comparisons can help identify how the distribution of habitat types differs across the ESU, as well as areas where specific types of habitats are rare (Figure 3-1a).

Barriers to Fish Passage

Data sets describing barriers to fish passage include natural barriers such as waterfalls and manmade barriers such as culverts, dams, and levees. Inventories are often available from state fisheries agencies, data cooperatives, transportation agencies, and federal land management agencies. A great deal of effort is often required to combine and validate existing data sets. In addition, one can expect a large number of barriers to be listed as “passage unknown” or “passable at times.” To know how to deal with such data, contact with local biologists and/or information describing the life-history strategy of the species of interest will be required. The major goal of a barrier analysis is to classify all stream segments as accessible or inaccessible. Inaccessible streams can be further delineated into naturally inaccessible versus those that have been cut-off by anthropogenic change. Areas that were historically inaccessible but which are now accessible present a challenge. Some uncertainty about whether some streams are accessible or inaccessible for a particular species will likely remain. This uncertainty can be quantified and reported to guide later field assessments. These analyses can help guide recovery planning by answering questions about the amount, type, and distribution of habitat that is currently inaccessible to salmonids, and why it is inaccessible (Figure 3-2).

Land-Use and Landform Data

Spatial data detailing landform and land use are needed to link stream habitats to the larger landscape. These data include generalized land cover data available, for example, from the 1992 National Land Cover Data project, as published by the USGS. More current Landsat Thematic Mapper data for portions of the United States are available for purchase from the Multi-Resolution Land Characteristics 2000 project, with the compiled data set expected to be available in 2004. The 1:250,000 Land Use Land Cover data are freely available from the USGS. Acquiring and formatting these remotely and field-derived data is a difficult step in conducting ESU-wide habitat inventories. These data can be used with data describing the distribution of habitat types in the watershed to examine the potential effects of land use on habitat quantities and quality (Figure 3-1a).

Habitat Field Surveys

Data detailing fine-scale habitat attributes such as pools and riffles are rare but often available for parts on an ESU through state and federal land management agencies. These data, when digitized, can be used to develop models that link landscape-level data about landform and land use to instream habitat conditions. Models derived from the limited data that do exist can be extrapolated to areas having little or no empirical estimates of pool and riffle habitats (Figure 3-1a). Provided data are available, habitat attributes such as pool:riffle ratios might be compared among land use categories (Figure 3-1a). Such comparisons are an important step in evaluating how interactions between land use and aquatic habitats may impact fish populations, and in predicting fish population information from instream habitat quantities.

Hydromodification Inventories

Comprehensive hydromodification inventories are rare but extremely valuable. These inventories quantify changes in aquatic habitats that have resulted from riprip, channel straightening, levees and other forms of modifications. Such observed data can be used to characterize the extent and location of modifications within watersheds and across ESUs. Furthermore, such data can be used to build models of predicted hydromodifications for areas where no surveys exist. Field surveys of riprapped banks, for example, might be used to calibrate a model that predicts probability of riprap from such variables as distance to the nearest road and channel width. Once hydro-modifications have been characterized throughout a watershed or ESU, these data can be linked with available fish and land-use data. Ultimately, these characterizations of observed or predicted hydromodifications will enhance our ability to predict fish population information from instream habitat quantities.

Connecting Instream Habitats to Fish Populations

One major challenge of recovery analyses is to identify whether changes in habitat have altered the abundance, productivity, diversity and spatial structure of salmon ESUs (Table 2-1). Having identified quantities of differing types of habitat available to salmonids using the ESU inventory, we can then broadly estimate the ability of habitats within an ESU to support juvenile and adult salmonids (abundance and productivity). The same habitat data might be used to examine how the distribution of habitats across an ESU relates to the life histories of populations within that ESU (diversity and spatial structure). In areas where historical habitat reconstructions are available, comparisons between current and historical habitats are possible. Such comparisons of current versus historical habitat are one approach for identifying possible factors for decline. For example, in the Skagit River Basin, major losses from historical habitat estimates in coho summer and winter habitats (Beechie et al. 1994) were attributed to hydromodification associated with agriculture and urban land uses.

In general, assessing the ability of habitats to support salmon populations involves associating fish density, use, and survival with individual habitat types (Figure 3-1c and see examples in Chapter 4). Because comprehensive field measurements of fish density in specific habitat types are generally not available for an entire ESU, compiled field and literature values of fish density in a given habitat type are needed. Having multiple measurements of fish density in a given habitat type allows us to bracket our uncertainty in estimating production or capacity by estimating capacity or productivity as a range of values derived from these compiled field and literature values.

Data needed to estimate capacity of habitats to support salmonids include 1) fish distribution (juvenile and adult), 2) field data on fish density and survival in specific habitat types (i.e., juvenile density in tributaries, survival estimates of juveniles in early rearing habitats, spatially explicit spawner surveys, etc.), 3) measured or predicted fine-scale habitat characteristics such as pool:riffle:glide ratios, and 4) area of lateral habitat (from ESU-inventory). "Fish distribution" is defined by identifying the areas within the ESU where adults spawn and juvenile fish rear. Data used to determine fish distribution include field observations when available, and information on barriers and stream gradient. In an ideal world, estimates of the ability of a given watershed to support salmon would be based upon field measurements of fish density within that watershed. As long as such data are not comprehensively available, analyses can make use of field data from multiple watersheds and compiled literature values.

Conclusion

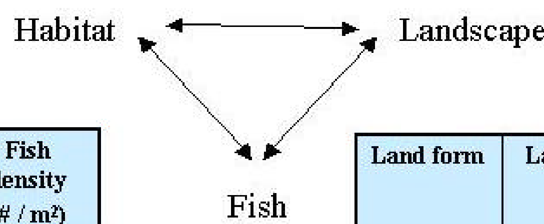
There are several advantages to conducting broad-scale analyses of ESU habitats. First, they allow us to efficiently examine large geographic areas using the same, consistent methodology. Because of this, we are able to compare results for different geographic areas within ESUs, clearly one of the major strengths of this method. Second, these coarse-scale analyses are driven by available data and require limited fine-scale data that are not readily available throughout broad geographic areas. Because data are limited, broad-scale analyses make use of models or extrapolations of data. Field validations will be needed to check how well these models and extrapolations reflect reality. Third, these data-driven analyses permit us to quantify uncertainty (see Chapter 8). And fourth, by comparing habitats across entire ESUs, these analyses can help focus recovery planning efforts and identify important areas of research.

The results of these analyses can form a foundation for future analyses, which undoubtedly must include finer-scale analyses evaluating habitat quality and function. One key challenge is how to scale between these fine- and coarse-resolution approaches. Undoubtedly, there will be times when predictions and results differ; yet conducting analyses at both resolutions can help identify knowledge gaps, uncertainties, and the best resolution needed to answer important questions for salmon recovery.

*****FOR REVIEW ONLY*****

a.

		Habitat type			
		Main stem		Tributary	
		Distribution	Characteristics	Distribution	Characteristics
Land use category	Urban	20%	% riffles	10%	% lost
	Forested	30 %	% riffles	60%	% lost
	Agriculture	50 %	% riffles	30 %	% lost
	Total	100%		100%	



c.

Habitat type	Fish density (# / m²)
Main stem	12
Tributary	15
Estuary	20

b.

Land form	Land use	Fish density (# / m²)
Erosive geology	Urban Forested Ag	12 20 5
Non-erosive geology	Urban Forested Ag	15 2 8

Figure 3-1. Illustration of the kinds of relationships between habitats, landscape, and fish that broad-scale analyses can examine: Table A illustrates how habitat quantity and characteristics are distributed across different land uses; Table B illustrates how fish density in three land use categories differ in erosive vs. non-erosive geologic settings; and Table C describes how fish density differs across habitat types. The numbers in the tables are completely fictitious and for illustrative purposes only.

	Total	Inaccessible	Accessible	Habitat Loss
Population A	26	8	18	0.308
Population B	9	6	3	0.667
Population C	17	5	12	0.294
Population D	15	0	15	0
Population E	20	4	16	0.2

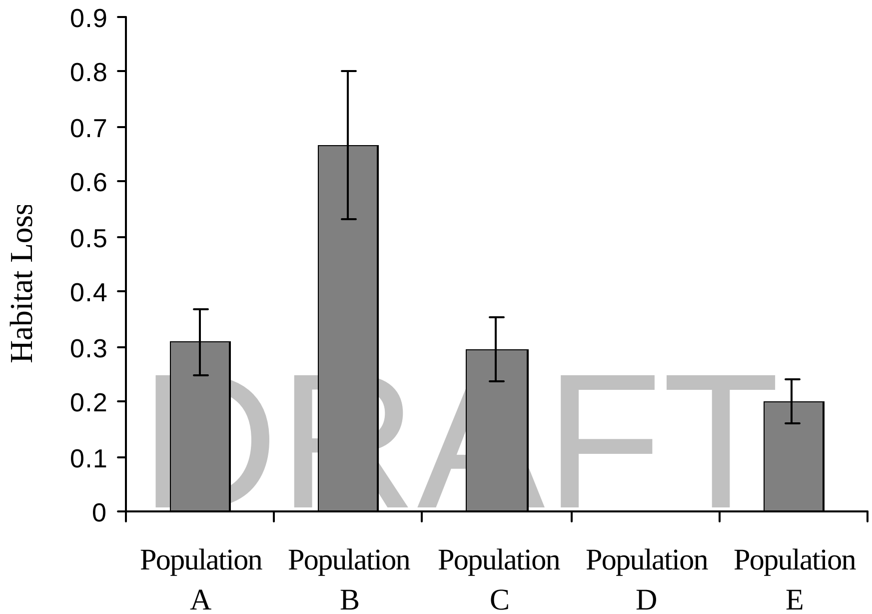


Figure 3.2. Example of possible results for an inventory of accessible and inaccessible habitats. Potential units for such an analysis include total stream miles, estimated number of pools (100s), m^2 off-channel habitat (1,000s), or stream miles of a particular habitat type, such as stream miles suitable for spring chinook spawning.

CHAPTER 4. ASSESSMENTS FOR IDENTIFYING AND PRIORITIZING RESTORATION ACTIONS

Timothy J. Beechie, George R. Pess, and Sarah Morley

In this chapter we describe a number of inventories and assessments that can help answer three important questions in salmon recovery planning:

1. How have ecosystem processes and functions that support salmon been disrupted by land uses?
2. What are historic and current habitat abundances for watersheds containing one or more genetically distinct salmon populations?
3. How have salmon populations likely been altered by changes in ecosystems and habitat conditions?

The first question is driven by the need to identify how the ecosystems that support salmon have been degraded and contributed to the decline of salmon populations. There are two groups of inventories required to identify ecosystem disruptions: identifying altered ecosystem processes and identifying impaired biological integrity. The first group of inventories describes historical and current ecosystem processes and functions (e.g., sediment supply, riparian functions, habitat connectivity) in order to determine which of them have been disrupted. The second set of inventories identifies locations where biological integrity has been impaired, and can indicate potential ecosystem disturbances that process assessments may miss.

The second and third questions are closely related, and are motivated by the need to estimate population goals for salmon stocks within an ESU. In Puget Sound, the TRT for threatened chinook salmon first estimates population goals by population viability analyses, which do not account for availability of habitat. The habitat-based method described here provides an independent estimate of the historical spawner abundance or smolt production, which the TRT can use for comparison with estimates made by population viability analysis. Assessing both the historical and current conditions within a watershed help in addressing these two questions. Contrasting the historical and current conditions provides an indication of the types of habitats most impacted and where those impacts have been greatest, which is useful for prioritizing habitat restoration actions identified by the ecosystem assessments. In aggregate, these three inventory and assessment questions provide a list of restoration actions needed to support salmon recovery (as well as to address the CWA), and provide the information needed to ascertain which actions will have the greatest impact on recovery of local salmon populations.

Assessing Degradation of Ecosystem Processes and Functions

Figure 4-1 is a conceptual diagram illustrating how watershed controls (ultimate and proximate) and natural landscape processes combine to form various habitat conditions. Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas (>1 km^2), and shape the range of possible habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (i.e., decades), act over smaller areas than ultimate controls, and are partly a function of ultimate controls (Naiman et al. 1992).

Landscape processes are typically measured as rates and characterize what ecosystems or components of ecosystems do. For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time) at which sediment or water is supplied to and transported through specific locations of a watershed. Certain riparian functions can be viewed similarly. For example, wood recruitment to streams from riparian forests and wood depletion from the channel are both rate functions. Natural rates of landscape processes are here defined as those that existed prior to non-Native American settlement and development activities, mainly forestry, agriculture, and rural and urban residential and commercial development.

Ecosystem processes and functions to inventory should include (at a minimum) hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality (Table 4-1). This suite of inventories is based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes. The list may not include all impacts to salmon in a watershed, but it includes those that are clearly supported by scientific literature and that are responsible for a significant proportion of the total loss in salmon production from Pacific Northwest river basins (e.g., Meehan 1991, Beechie et al. 1994, Beechie et al. 2001). For each process a series of diagnostics can be developed based on rates from scientific literature and local studies.

These inventories identify 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and habitat conditions. Habitat restoration and protection actions resulting from these assessments are directed at protecting and restoring beneficial habitat-forming processes instead of attempting to build specific habitat conditions (Beechie and Bolton 1999). These assessments systematically identify land use disruptions to habitat-forming processes at two levels of resolution. Coarser level assessments locate disturbed habitat-forming processes using a combination of Geographic Information System (GIS) data and field-based inventories to identify alterations to peak flows, sediment supply, riparian conditions, blockages to salmon migration, water quality, and channel and floodplain interactions. These assessments provide broad-brush screening tools for understanding where processes are disrupted, and in some cases for estimating total costs of restoration actions. The finer level assessment relies solely on field-based inventories and identifies specific restoration or protection actions that are required for recovery.

Inventories of ecosystem processes and functions can be grouped into three categories: 1) distributed watershed processes (similar to non-point sources, such as sediment supply), 2) reach-level processes that primarily affect the adjacent reach (e.g., riparian functions), and 3) other ecosystem functions (e.g., habitat connectivity). The inventory of disruptions to distributed watershed processes has two levels of resolution: a coarse resolution that identifies areas within a watershed where land uses have increased rates above natural background levels, and a detailed field inventory that identifies specific areas where restoration actions are needed. Inventories of disruptions to reach-level processes can also have two levels of resolution: a coarse resolution analysis that broadly indicates patterns of degradation in the watershed, and a detailed field inventory that identifies site-specific disruptions to reach-level processes and potential restoration actions. Other ecosystem functions are necessary attributes of ecosystems that are not readily analyzed as rates or levels of function. For example, barrier inventories describe where habitats have been blocked to salmon access, and flow diversion inventories describe direct alteration of flow regimes at multiple points. Neither type of assessment neatly fits into the categories of watershed-level or reach-level processes.

Watershed-level Processes

Watershed level processes are those that have multiple, widely distributed sources, including sediment supply, hydrology, and inputs of nutrients or pesticides (Table 4-1). Describing how these processes have been disrupted and what restoration actions are required for their recovery requires two different kinds of assessments. First, process assessments identify the degree to which process has been altered by land use, and where in each watershed these changes have occurred. Second, inventories are required to identify where specific restoration actions must be taken in order for recovery to occur.

Identifying altered watershed processes

We present two main approaches to understanding disruptions to these processes: budgeting and landscape indicators based on known relationships of certain land uses to the parameter in question. The budgeting approach is most often used for sediment supply and inputs of nutrients or pesticides to water bodies. The general budget can be stated in equation form:

$$\Delta S = I - O$$

Where, ΔS is change in storage, I is input, and O is output. In essence, S is the stream condition for any parameter (e.g., the amount of sediment or of a pesticide in the stream), and quantifying changes in inputs or outputs indicates how land uses have altered the stream ecosystem. In many cases, it may only be necessary to quantify how inputs have been altered by land uses, which is called a partial budget. That is, where changes to outputs are negligible, an increased input is equal the change in storage and to the altered stream condition. Therefore, it is not necessary to understand output processes in detail (e.g., sediment transport) in order calculate change in storage and to understand how the stream ecosystem has been altered.

Landscape and land use indicators of changes to watershed processes can also make valuable assessment tools. Where there are known relationships between land uses and changes to process rates (e.g.,), simple screens can be developed to indicate where specific processes have likely been altered. For example, known relationships among zoning, impervious surface area, and changes in hydrologic processes and biota have been used in indicate changes hydrologic regime in urban areas (see Appendix A). Where impervious surface areas are less than 3% of the watershed area, hydrologic regime is not significantly different from one with no impervious surfaces (Booth and Jackson 1997). However, where impervious surfaces are more than 10% of the watershed area, hydrologic regime has likely been altered to the point where changes in biota are severe. Similar coarse screens can be developed for other watershed processes such as pesticide runoff from agricultural areas.

For any watershed process, elements of the two approaches may be used in tandem to make best use of limited field information and to focus field efforts in areas with a high likelihood of disruption. Limited field data can be used to generate statistical relationships among landscape or land use factors and the parameter being assessed, and these correlations can then be used to identify areas of the landscape processes are likely. For example, partial sediment budgets for 10 sub-basins in the Skagit basin (covering approximately 10% of the watershed area) quantified sediment produced by landslides since the 1940s using the aerial photograph record (Paulson 1997). From these data, Skagit System Cooperative compiled average sediment supply rates by lithologic group and land use in order to estimate average sediment supplies for the remaining watershed area (see Appendix A). These average sediment supplies were then mapped in the rest of the watershed using GIS, and average sediment supplies were calculated for each sub-watershed area in the basin. This coarse screen indicates where average sediment supply from landsliding has likely increased more than 50% over natural rates, and where restoration or rehabilitation may be necessary.

Identifying restoration actions for watershed processes

There are two types of restoration actions that may be taken to restore watershed processes: active and passive. Passive restoration actions are those where a land use is removed or altered in such a way that a process recovers naturally. Active restoration projects are those where specific interventions are used to assist recovery of a watershed process. In general, remote sensing methods and mapping are used to identify areas for passive restoration, and field inventories are required to identify active restoration projects. For example, mapping of landslide hazard areas is used to identify areas that are particularly prone to landsliding, and sensitive to land uses such as clearcut logging or road building (Figure 4-2). Such maps are tools for passive restoration, which allows recovery sediment supply rates by preventing or modifying land uses within hazard areas. Second, inventory of road landslide hazards identifies specific areas for active restoration. Road inventories should identify segments of road that are at risk of failure (e.g., Renison 1998), as well as specific stream crossings, cross drains, or fills that are likely to fail. Each potential failure site can be itemized on project lists for restoration action. The restoration actions can then be prioritized based on potential impact to stream habitat, smolt production, and cost (see Chapter 5).

Reach-level Processes

Reach-level processes are those processes that directly affect the reach of interest and where the most direct linkages between land uses and habitat alterations are local (Table 4-1). These processes mainly include riparian functions and floodplain-channel interactions. An extensive body of literature describes linkages between riparian forest functions and stream habitat, which in turn affect the productivity and abundance of salmonids. Riparian functions include supply of wood and leaf litter to streams, shading, and root reinforcement of stream banks and floodplain soils. For this example we focus on recruitment of wood to streams and its function in channels, which are among the most studied of riparian functions (e.g., Murphy and Koski 1989, Bilby and Ward 1991, Montgomery et al. 1995, Abbe and Montgomery 1996, Beechie and Sibley 1997). The level of wood input or other riparian functions increases with increasing width of forest buffer on streams (Figure 4-3), and the proportion of the function occurring within a given distance of the channel edge varies by function (Sedell et al. 1997). These relationships can be used to evaluate the current status of functional interaction between a stream reach and riparian area, and indicate whether existing levels of riparian protection are sufficient to ensure continued function.

Floodplain-channel interactions and their effects on habitat are less clear in the literature. However, effects of riprapping and leveeing of rivers on isolating habitats and the magnitude of those losses has been documented (e.g., Beechie et al.1994, Beechie et al. 2001). Methods for detecting and quantifying such changes are straightforward and relatively inexpensive (see Appendix A). Effects on land uses on the dynamics of rivers is also clear, but the resulting effects on habitat abundance and quality are not. Therefore, we do not suggest extensive analyses of changes to river dynamics at this time.

As with watershed level processes, there are two types of assessments required for reach-level processes. The first assessment identifies where reach level processes have been disrupted. The distribution of riparian conditions at this larger spatial scale can provide a general sense of the change in riparian function from historic conditions (e.g., Lunetta et al. 1997). Subwatersheds where the current distribution of riparian conditions deviates markedly from that expected under a natural disturbance regime are locations where riparian restoration efforts may be appropriate. The same data can also help managers understand how different land use practices differ in their degree of impact on riparian functions. These relationships can then help assess the potential impacts of large-scale land use policies on salmon habitat recovery (e.g., evaluating potential effects of growth management legislation).

Because of limitations in the satellite classification of riparian forests, field inventories of riparian sites must be used to identify specific restoration actions. Field inventories may consist of initial measurements and classification from aerial photography, combined with field confirmation of the riparian vegetation conditions for each stream reach. At a minimum, they should classify riparian conditions by buffer width, stand type, and age of vegetation. From the data, managers can identify impaired or moderately impaired stream segments (i.e., those with forest buffers <40 m wide), determine the likely cause of that impairment, and identify required restoration actions.

Regardless of current condition of riparian areas, establishing protected areas along the channel where natural riparian vegetation can develop through time and interact with the stream is a necessary component of riparian restoration. Active restoration efforts may be appropriate at currently impaired sites. Riparian restoration may include the planting of desired riparian plant species or manipulation of the existing vegetation to accelerate tree growth and the development of desired stand structural characteristics (Berg et al. 1996, Beechie et al. 2000).

Other Ecosystem Functions

[Here we will discuss several examples of inventory of blocked tributary habitat. We will also put in another example besides the Skagit, add more information about the criteria used to determine blockages, and discuss.]

Impaired fish passage

Stream crossing structures that block fish access to useable habitats can account for as much as 50% of lost smolt production from tributaries in Puget Sound river basins (Beechie et al. 1994, Beechie et al. 2001). Assessing such isolation of habitats is one of the simplest inventories that can be conducted because criteria for fish migration blockages are relatively clear and identifying the amount of habitat affected involves little subjectivity. Moreover, combining these inventory results with cost estimates for restoration actions allows managers to rank the cost-effectiveness of individual projects in order to more effectively direct the expenditure of limited restoration funds.

The Stillaguamish and Tulalip Tribes used such an inventory to identify isolated habitat projects in the Stillaguamish River basin (Figure 4-4 [need to insert it]), following the Fish Passage Barrier and Prioritization Manual of Washington Department of Fish and Wildlife (WDFW 1998). Based on inventory of 500 structures as of 1998, they identified 50 structures that do not meet passage criteria on tributaries, ___ that require a more in-depth hydraulic analysis to determine whether or not they meet the criteria, and ___ that meet passage criteria (Stevenson and Griffith, unpublished data). The remaining ___ structures, mainly those on side channels or distributary channels that are currently disconnected from the river, were labeled "unknown" because passage cannot be evaluated until designs for reconnecting river flows to these channels have been developed.

Using this inventory the Stillaguamish and Tulalip Tribes evaluated the cost-effectiveness of projects based on the habitat area upstream of the project, multiplied by the average life span of a blockage (~50 years) and divided by the cost of the project. These results allowed the tribes to identify the most cost effective projects for reconnecting blocked tributary habitats based on benefits to multiple salmonid species, as well as costs of reconstructing individual stream crossings.

Water diversions

[Michelle's section]

Assessing Biological Integrity

A key component of salmon habitat is the stream biota itself. Invertebrates, amphibians, diatoms, and other stream organisms are integral parts of the aquatic food web upon which endangered fish species depend. These assemblages are also sensitive to a variety of watershed disturbances expressed over multiple spatial scales, and therefore excellent indicators of stream condition. Unlike anadromous fishes that are subject to a wide variety of human disturbances in the marine and freshwater environment (e.g., migration blockages, interaction with hatchery fish, damaged estuarine habitats, or oceanic overharvest) less migratory stream organisms (such as benthic invertebrates) often provide a more accurate reflection of site condition. Over the past century, biological assessment (measuring and evaluating biota directly) has ranged from saprobien indexes (Hilsenhoff 1982), to toxicity testing (Buikema and Voshell 1993), indicator species abundance (Farwell et al. 1999), diversity indexes (Wilhm and Dorris 1966), and more recently to multivariate models (Wright et al. 2000) and multimetric indexes (Davis and Simon 1995).

Multivariate models

In this approach a predictive model is developed based on a large (≈ 200 sites) data set of reference (minimally disturbed) sites (Reynoldson et al. 2001). Using multivariate statistical analyses, references sites are matched to a set of habitat descriptors (e.g., stream order, elevation, etc.) and classified into groups. Level of impairment at a given sample site is then determined by comparison to the appropriate reference group. This approach has been most widely applied in England with the development of RIVPACS (River Invertebrate Prediction and Classification System), in Australia with AUSRIVAS (Australian River Assessment Scheme), and with BEAST (Benthic Assessment of Sediment) in Canada (Wright et al. 2000). In the Pacific Northwest, multivariate models have been developed in British Columbia for the Fraser River basin (Reynoldson et al. 2001) and in Oregon with the BORIS (Benthic Evaluation of Oregon Rivers) model (Canale 1999). Based on benthic invertebrates, BORIS scores a site from 0 (severe impairment) to 100 (comparable to reference condition). A RIVPACS-type predictive model applicable to wadeable streams throughout western Oregon and Washington is currently being developed (Hawkins and Ostermiller 2001).

Multimetric indexes

Multimetric indexes, such as an index of biological integrity (IBI), integrate empirically tested attributes (metrics) of stream biotas (Karr and Chu 1999). This approach was first developed using fish communities in the midwestern United States (Karr et al. 1986), but has since been modified for a variety of assemblages—most commonly fish (Simon 1998), invertebrates (Kerans and Karr 1994), and algae (Hill et al. 2000). At the national level, the rapid bioassessment protocols used by the U.S. Environmental Protection Agency are based in a multimetric approach (Barbour et al. 1999). As with multivariate models, IBIs and other multimetric indexes are regionally calibrated based on ecoregion designations and local reference conditions. In the Pacific Northwest, an IBI using benthic macroinvertebrates has been calibrated with data from the Puget Sound lowlands of Washington (Kleindl 1995, Morley 2000), and from southwestern (Fore et al. 1996) and northwestern Oregon (Adams 2001). This index, known as the benthic index of biological integrity or B-IBI (Karr & Chu 1999), is composed of ten measures of taxa richness, population structure, disturbance tolerance, and feeding ecology (Table 4-2). When scores from these metrics are summed, B-IBI provides a numeric synthesis of site condition that ranges from 10 (poor) to 50 (excellent), and can determine five categories of resource condition (Doberstein et al. 2000).

Applications

State regulatory agencies in Washington (Department of Ecology) and Oregon (Department of Environmental Quality) both currently use a combination of multimetric and multivariate approaches to assess the condition of streams and rivers (Mochan and Mrazik 2000, Plotnikoff and Wiseman 2001). Instead of focusing on the response of only one species to habitat alteration, bioassessment protocols such as B-IBI or BORIS provide integrative ecological measures of stream condition that are based directly on the organisms that live there (Angermeier 1997). As such, these bioassessment tools can be used as measuring sticks to characterize the level of impairment in a stream basin, identify causes of degradation, evaluate the effectiveness of restoration actions, and identify areas of high biological integrity for conservation (Barbour et al. 1999, Morley and Karr in press).

In the Puget Sound region, B-IBI has been applied by city and county agencies (King County 1996, Thornburgh and Williams 2000), university scientists (May et al. 1997, Larson et al. 2001, Morley and Karr in press), and volunteers (Fore et al. 2001) to report on the biological condition of surface waters, to screen watersheds for further physical or chemical monitoring, and to evaluate various restoration and conservation strategies. In Washington State, B-IBI has been used to evaluate wood placement in both forested (O'Neal et al. 1999) and urban basins (Larson et al. 2001). Studies done at the University of Washington have examined how B-IBI responds along a gradient of human disturbance—be it forestry (Fore et al. 1996), urbanization (Morley and Karr in press), or recreation (Patterson 1996)—knowledge that allows managers to identify critical areas for protection or further monitoring. Water resource agencies such as Snohomish County Surface Water Management in Washington State currently uses B-IBI in conjunction with chemical and physical data to track the health of their streams over time (Thornburgh and Williams 2000).

Tuning restoration efforts to site-specific needs is also enhanced by using biology to aid in the detection of the primary causes of degradation. Multimetric indexes such as B-IBI provide a numeric synthesis of the biological dimensions of site condition, but they can also be broken down to derive descriptive and potentially diagnostic information from each of the component metrics (Karr et al. 1986). In Washington State alone there are hundreds of species of stream invertebrates—each with specific life-history requirements and varying tolerance to specific forms of disturbance (Rosenberg and Resh 1993, Merritt and Cummins 1996). Stoneflies, for example, generally require cool, well-oxygenated waters while many invertebrates that are classified as "clingers" prefer stable and sediment-free substrate (Merrit and Cummins 1996). In urban streams of the Puget lowlands, Morley and Karr (in press) found a positive correlation between clinger taxa richness and measures of substrate size distribution. Other studies have indicated that mayflies may be excellent indicators of heavy metal contamination (Kiffney and Clements 1994, 1996, Kleindl 1995). Further investigation of the relationships between specific metrics of B-IBI and particular effects of human disturbance is needed. Knowledge of these relationships could be an important guide to diagnosing the site-specific causes of degradation.

Estimating Current and Historical Potential Fish Production

Watershed assessments for estimating current and historical habitat availability and production potential require associating seasonal fish use and survival with a habitat type (Reeves et al. 1989, Beechie et al. 1994), and quantifying areas of different habitat types. Quantifying fish use of different habitat types requires specific studies designed to identify these relationships (e.g., Bisson et al. 1988). However, it may not be necessary to develop new relationships between fish use and habitat types in each watershed if it can be demonstrated that fish use and comparable habitat types are similar to those in previous studies. For example, different juvenile Pacific salmon species spatially segregate into different habitats in a watershed, as illustrated in Figure 4-5 (Pess et al. in press). Steelhead (*Oncorhynchus mykiss*), coho (*O. kisutch*), and ocean-type chinook (*O. tshawytscha*) have higher densities in specific habitat types for the same life stages. Juvenile steelhead utilize all habitat types in a similar fashion, while juvenile coho have greater preference towards slower water habitat such as side channels and ponds, as evidenced by higher densities in those areas. Ocean-type juvenile chinook prefer mainstem and estuary environments. Adult Pacific salmon are also temporally and spatially segregated during the spawning life stage (Lichatowich 1999, Montgomery et al. 1999).

Unlike fish-habitat type associations, habitat data are not transferable across watersheds because the natural potential and effects of land use vary by watershed (Lunetta et al. 1997, Beechie et al. 2001, Collins and Montgomery 2001). Therefore, habitat inventories must be conducted separately in each river basin or planning area. Assessments for estimating historical and current habitat are conducted in three steps: 1) identify habitat types, 2) estimate historical and current habitat abundance by type, and 3) estimate production potential based on habitat-fish relationships.

Identify Habitat Types

A habitat classification system suitable for estimating historical and current habitat and potential fish production must have two main attributes. First, analysts must be able to associate fish abundance and survival with each habitat type in order to estimate total fish abundance in a watershed. Second, it must be possible to quantify historical and current habitat areas in order to estimate changes in potential production over time. We recommend a suite of habitat types at two hierarchical scales, coarse and fine (Table 4-3). The coarser resolution of habitat types can be mapped from remotely sensed data at the reach scale (e.g., topographic maps, aerial photography, or satellite information), whereas the finer resolution of habitat types must be identified in the field at the habitat-unit scale (sometimes with the aid of aerial photography). Because these typing systems are nested, all reaches within a watershed can be stratified by landscape and land use factors using the remotely sensed coarse-resolution data, and reaches within each strata can be subsampled to develop an understanding of habitat types within each reach type. This hierarchical relationship enables extrapolation of habitat conditions for unsampled reaches within the watershed.

Stratification of reach types may include several different landscape and land use factors, although a relatively small number of strata are desirable to reduce the complexity and number of assumptions and calculations. For example, tributary reaches may be stratified simply by slope and land use in order to identify changes in pool area as a result of land uses (Beechie et al. 2001). This stratification is useful because reaches of different slope have different natural potentials for pool formation (e.g., Montgomery and Buffington 1997), and reduced wood abundance (a common impact of land use) has a more pronounced effect on pool frequency and area in certain slope classes (e.g., Montgomery et al. 1995, Beechie and Sibley 1997). The same slope classes are not particularly relevant for large rivers, where some combination of slope and discharge may be more useful in predicting natural channel patterns (e.g., Leopold et al. 1964). Stratification by potential natural channel pattern and land use can be used to help identify where river and floodplain interactions have been altered, and therefore where changes in the finer-resolution habitat types have occurred.

Estimate Current and Historical Habitat Abundance by Type

Methods for estimating current and historical habitat abundance differ among habitat types. For example, estimated changes in tributary pool areas are based primarily on data from reference sites within the study area (Beechie et al. 1994), whereas estimated changes in channel and wetland areas on deltas are primarily from maps and survey notes made prior to extensive settlement by non-Native Americans in the mid 1800s (Collins and Montgomery 2001). Therefore, it is not possible to describe a single methodology for assessing changes for all reach types. Instead, we provide an overview of different methods that one might use for assessing habitat conditions historically and at present. More detail on individual methods can be found in Beechie et al. (1994), WDFW (1998), and Collins and Montgomery (2001).

Reduction of pool areas in tributary habitats can be estimated by comparing pool areas in streams impacted by land use to pool areas in reference streams relatively unimpacted by human activities (Beechie et al. 1994). Even in watersheds unaffected by logging or other land use, factors such as wildfire, debris torrents, or floods introduce variability in habitat conditions among reaches (Bisson et al. 1997). Therefore, enough reach types in each stratum should be sampled in both impacted and reference watersheds to assess the variability among reaches in the relative abundance of different habitat types. This variability can also be reduced by ensuring that reference streams are as similar as possible to those in impacted areas with respect to geomorphic setting and potential natural vegetation. That is, stratification of reach types for sampling will help limit variability within reach types and increase chances of detecting differences between impacted and unimpacted reaches. A similar approach can be used for larger rivers (e.g., >30 m wide), although processes of habitat formation and the range of habitat types in these rivers differ from those in tributaries, and the availability of appropriate reference reaches may be limited (Collins and Montgomery 2001).

A second approach for assessing habitat changes in larger channels uses historical records and maps to reconstruct historical habitat conditions. Historical areas of slough habitats (both side channels and distributaries) can be estimated from historical maps, notes, and photos, and often can be field verified by residual evidence of their prior locations (Beechie et al. 1994, Collins and Montgomery 2001). Present day areas can be measured from aerial photographs and in the field. Comparisons of the two inventories identify areas of significant habitat loss and suggest how land uses have changed habitat-forming processes. Similar methods are used to identify losses of off-channel habitats on floodplains.

Changes to lake areas are measured directly from historical and current maps (Beechie et al. 1994), and typically indicate where rivers have been dammed for hydropower or water supplies. Lake areas tend to increase, whereas mainstem and tributary habitats are inundated by reservoirs and decrease in extent. Pre-settlement beaver pond areas can be estimated based on frequencies of beaver ponds in relatively pristine areas (e.g., Naiman et al. 1988), and present-day pond areas within the study area can be measured using field surveys and aerial photography. Losses of beaver pond habitat are then estimated by comparing estimates of historical and current areas (Pollock and Pess 1998).

Portions of tributaries that are no longer accessible to salmon can be mapped using inventories of habitat upstream of migration barriers. Natural barriers to salmon migration must first be identified to delineate the assessment area. All structures crossing streams within the assessment area (culverts, bridges, small dams) should then be inventoried to determine if they meet passage criteria for salmon. Finally, habitat areas upstream of each manmade barrier must be surveyed to determine how much habitat is inaccessible (i.e., that area upstream of the identified barrier and downstream of the natural barrier to salmon migration). Use of a standardized method for determining blockages (e.g., WDFW 1998) will streamline identification and prioritization of isolated habitats and provide a standard set of criteria for monitoring progress toward reopening these habitats.

Estimate Production Potential Based on Habitat-Fish Relationships

At its simplest, smolt production potential from a given habitat type or area is calculated as:

habitat area × average fish density × survival to smolt. (1)

However, comparing the impact of different habitat alterations on smolt production potential requires making separate estimates for each habitat type. Thus, the production potential of a habitat for each life stage (e.g., spawning, egg to fry, summer rearing, winter rearing, smolt migration) can be expressed mathematically as:

$$N = \left(\sum_{i=1}^{n} \left(\left(\sum_{j=1}^{n} A_{ij} \right) \times d_i \right) \right) \tag{2}$$

where ΣA_{ij} is the sum of areas of all habitat units (j =1 through n) of type i, and d_i is the density of fish in habitat type i. To compare capacities among life stages and identify which habitats may be limiting smolt production, the population estimate (N) for each life stage in a given habitat is multiplied by density independent survival to smolt stage so the capacities can be compared in terms of number of smolts ultimately produced (Reeves et al. 1989). Equation 2 can also be used to estimate historical spawner capacity based on estimates of historical habitat availability. Both spawning and rearing capacities can then be incorporated into assessments of factors that limit population size.

Examples of Application

Two examples of this assessment procedure are a historical assessment of coho salmon rearing habitat and smolt production losses in the Skagit River basin (Beechie et al. 1994), and a similar unpublished study for the Stillaguamish River basin (Pess et al. 1999b). In these assessments, inventories of current and historical habitat types and abundance provide the basis for estimating historical and current smolt production. Both studies used habitat-specific rearing densities and average seasonal survivals from Reeves et al. (1989) to estimate historical and current coho salmon smolt production, evaluate the effects of different land uses on coho production, and identify recovery of lost off-channel habitats such as sloughs and ponds as a necessary focus for any recovery effort intended to significantly increase coho smolt production (Beechie et al. 2001). A similar effort to estimate population responses of ocean-type chinook salmon to habitat change is currently underway in the Skagit River basin. Because of the importance of estuary rearing to ocean-type chinook, the assessment will also include relationships developed for estuarine rearing habitat.

Conclusions

Assessments of the current and historical conditions of a watershed can greatly improve our efforts to plan, implement, and monitor habitat restoration for the recovery of Pacific salmon. Systematically collected habitat data, a more thorough understanding of fish responses to habitat change, and a greater understanding of stream biota will allow refinement of the modeling tools used to predict fish and other biological response from application of different restoration strategies. These refinements will improve estimates of rates and pathways of recovery for many salmonid species in Puget Sound rivers, and assist in prioritizing restoration actions. However, many of these refinements are still several years from completion.

In the interim, systematic inventories of disrupted habitat-forming processes and blockages to salmon migration should be conducted to provide a complete river basin overview of necessary restoration actions that can be prioritized and sequenced logically. A minimum set of inventories for Puget Sound river basins should include barrier inventories, landslide inventories, floodplain and riparian characterization, channel and valley type classification, road, and biological indicator inventories. Some of these data are already available for parts of many watersheds in Puget Sound. These data provide the basis for identifying needed restoration actions, which can be prioritized by cost-effectiveness, influence on particular species, adjacency to existing centers of biological productivity or diversity (commonly referred to as: refugia, biological hot spots, source watersheds, core areas, key habitat), or other strategies.

There are many sources of uncertainty in these assessments, as well as in the political landscape surrounding restoration of Puget Sound watersheds. Uncertainties in the two assessments stem from natural variability in habitat-forming processes, habitat characteristics, and fish populations, as well as from errors in assumptions and limitations of data or knowledge. Our ability to characterize these types of uncertainty is limited by availability of data on watershed processes, habitat conditions, and fish populations over long periods of time. Lack of knowledge about current habitat conditions or responses of fish populations to changing habitat conditions introduce uncertainty into predictions of fish responses to watershed and habitat restoration. As with any model, improving the quality of the data reduces uncertainty related to knowledge gaps and improves ability to address the uncertainty related to natural variability in fish response to habitat conditions.

The changing nature of the political landscape also introduces uncertainty into recovery planning. Political uncertainties include potential changes in the species of concern (or listing of additional species in the future), and shifting public policy affecting regulatory protection of salmon habitat and funding for restoration. In the early 1990s, management of salmon habitat in Puget Sound was driven largely by interest in coho salmon, which limited fishing on many stocks under weak-stock management policies. The listing of Puget Sound chinook salmon as threatened under the ESA in 1999 dramatically shifted the emphasis of habitat studies and restoration efforts. Potential future listings of other species may further complicate recovery planning because the habitat requirements of different species will not match those of chinook salmon. Therefore, prioritizing restoration actions for single species may create additional conflict in recovery planning.

Recovery plans designed to protect and recover processes that create and sustain riverine habitats in Puget Sound are more likely to recover salmon of all species, and help avoid future conflicts among species. Use of a comprehensive assessment process and developing restoration plans focused on the reestablishment of habitat-forming processes minimizes conflicts that can arise with species-centric restoration approaches. Restoration of habitat-forming processes targets restoration of the natural array of habitat types and conditions within a watershed, which is consistent with the concepts of watershed and ecosystem management supported by the scientific community. Moreover this approach focuses on the natural potential of each watershed, and therefore is most likely to restore the diversity and abundance of stocks appropriate to each watershed in Puget Sound.

Table 4-1. Examples of methods used for rating individual landscape processes**. (Need to generalize this entire table, and add east-side examples)**

Distributed watershed processes

Hydrology - peak flow

Lowland basins: Hydrologic impairment in lowland basins can be rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions. EIA ≤ 3% is considered “functioning”, EIA between 3% and 10% is “moderately impaired,” and EIA > 10% is “impaired” (based on Booth and Jackson 1997, and see example in Appendix A).

Mountain basins: Peak flow ratings for mountain sub-basins can be developed based on empirical correlations between land use and elevated peak flow in forested basins (Jones and Grant 1996, and see example in Appendix A).

Sediment supply

Estimating impairment of sediment supply: Changes in average sediment supply for forested sub-basins within a watershed can be estimated based on present-day sediment supply rates from unlogged, clearcut, and roaded portions of the watershed (Dietrich and Dunne 1978, Paulson 1997, Montgomery et al. 1998).

Surface erosion on agricultural range lands Use something like USLE for these.

Inventory - identify sediment reduction projects: Inventories must focus on factors that influence sediment supply, identification of landslide hazard areas so that forest practices can be avoided or modified in sensitive areas (e.g., WDNR [DATE?], Montgomery et al. 1998), such as risk of road-related landslides (e.g., Renison 1998), crop management practices that increase surface erosion (e.g., refs), or grazing practices that alter sediment supply. The final value, called the risk rating, ranks roads with respect the threat that they pose to salmon habitat. Higher risk ratings indicate greater chance that a road will fail and impact salmon habitat. Final ratings were grouped into three categories of risk. A rating >30 is high, 16 to 30 is moderate, and ≤15 is low.

Pesticide delivery?

Reach-level processes

Riparian function

Remote sensing assessment: Riparian forests that are > 40 m wide are considered “functioning.” Forested buffers 20 to 40 m wide are considered “moderately impaired.” Forested buffers < 20 m wide are considered “impaired.” The proportion of impaired, moderately impaired, and functioning riparian forests can be estimated using Landsat classifications of vegetation.

Field inventory: In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which gives sufficient information to prescribe generalized management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.

Channel and floodplain interactions

Floodplain areas were delineated where the 100-year floodplain was greater than two channel widths using Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs. Reach breaks were based on differences in floodplain width and changes in channel pattern. Hydromodified areas were delineated on aerial photos by rafting or jetboating each main channel within floodplain reaches.

Other ecosystem functions

Habitat connectivity
Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tide gates, bridges, dams, and other manmade structures). The inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids (e.g., WDFW 1998).

Altered flow regime and sediment supply by dams or diversions (generally not considered under the hydropower H)

Hydrology – water storage or withdrawal
Hydrograph records and synthetic hydrographs, statistical analysis of flow variables

Sediment supply
Routing estimates, accounting for attrition.

Diversion screening

Table 4-2. The 10 metrics of the benthic index of biological integrity (B-IBI), and their predicted response to increasing human disturbance.

Metric	Description	Response
Taxa richness and composition		
Total taxa	Richness	Decrease
Mayfly taxa	Richness	Decrease
Stonefly taxa	Richness	Decrease
Caddisfly taxa	Richness	Decrease
Population structure		
Dominance by top 3 taxa	Relative abundance	Increase
Long-lived taxa richness	Richness	Decrease
Tolerance and intolerance		
Intolerant taxa richness	Richness	Decrease
Tolerant taxa	Relative abundance	Increase
Feeding and other habits		
Clinger taxa richness	Richness	Decrease
Predators	Relative abundance	Decrease

*****FOR REVIEW ONLY*****

Table 4-3. Habitat types used for the two types of watershed assessments described in this paper. Coarser scale habitat types are mapped from topographic maps, aerial photography, and satellite imagery. Finer scale habitat types are mapped using a combination of aerial photography (for larger units) and field measurements.

Habitat type (Coarser scale)		**Habitat type (Finer scale)**

Large main stems (>50 m bfw) by channel type based on gradient and confinement	• Edge
	• Mid-channel
		• Pool
		• Glide
		• Riffle
		  Boulder/cobble
		  Cobble/gravel
		• Bar edge
		• Bank edge
		  Natural
		  Hardened
		• Backwater (alcove)

Small main stems (10-50 m bfw) and tributaries (<10 m bfw) by channel type based on gradient and confinement	• Pools
	• Riffles
		• Pool
		  Scour
		  Plunge
		  Trench
		  Backwater
		• Glide
		• Run
		• Rapid
		• Riffle

Off-channel habitat within large main channel floodplains	• Channel-like • Pond-like	
Impoundments		• Ponds < 500 m^2 • Ponds > 500 m^2 and < 5 ha • Lakes > 5 ha
Palustrine wetland	• Forested • Scrub/shrub	Open water area by season
Riverine tidal wetland	• Forested • Scrub/shrub	Open water area by season and tidal stage
Estuarine wetland	• Scrub/shrub • Emergent	Open water area by season and tidal stage
Estuarine channel	• Main stem • Blind • Distributary	

*****FOR REVIEW ONLY*****

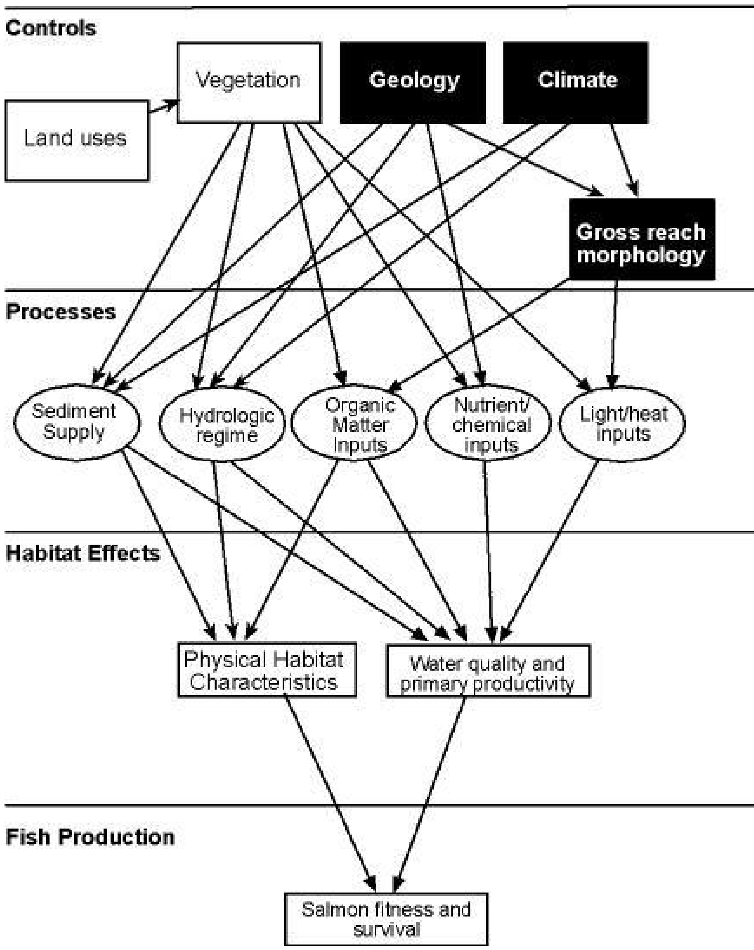


Figure 4-1. Schematic diagram of relationships between controls on watershed processes, effects on habitat conditions, and salmon survival and fitness (adapted from Beechie and Bolton 1999). Dark boxes in upper row are ultimate controls, light boxes are proximate controls.

*****FOR REVIEW ONLY*****

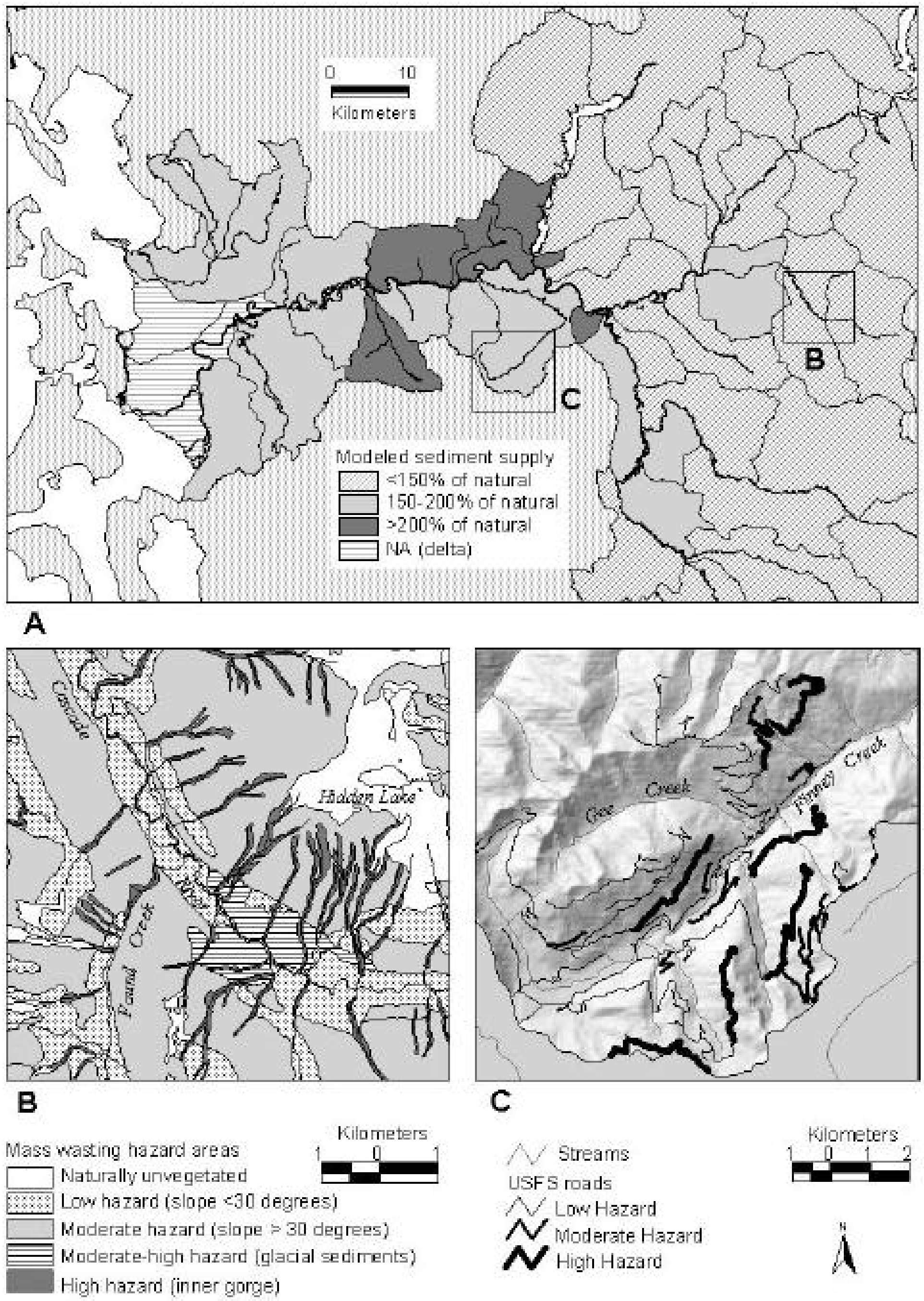


Figure 4-2. (A) Map of areas in the Skagit basin where sediment supply has likely increased due to land use, based on extrapolation of data from sediment budgets (described in text). (B) Landslide hazard map for a portion of the upper Cascade River basin. (C) Hazard map of U.S. Forest Service Roads classified as high risk of failure, moderate risk, or low risk.

*****FOR REVIEW ONLY*****

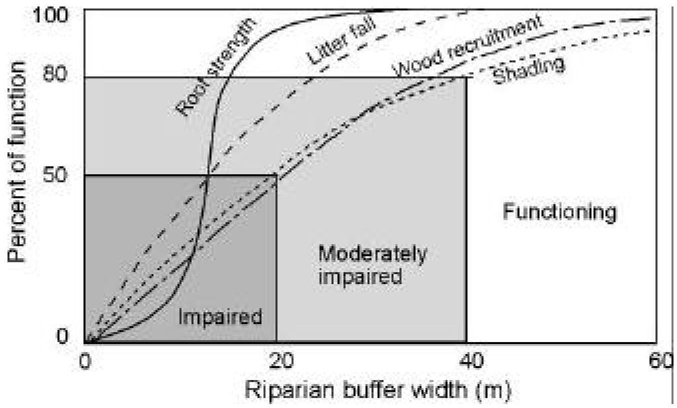


Figure 4-3. Illustration of change in riparian function with distance from channel (curves adapted from Sedell et al. 1997), and the Skagit Watershed Council's classification of impaired, moderately impaired, and functioning riparian forests.

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[Need to insert figure here]

Figure 4-4. Example map of inventoried stream crossing structures for a portion of the Stillaguamish River basin (enlarged area), showing structures that are blocking, passable, or needing further analysis. Dark gray area in watershed map indicates the historical extent of anadromous fish access based on mapping of ___ natural barriers. All stream crossing structures within the historical anadromous zone (more than ___ total) were inventoried.

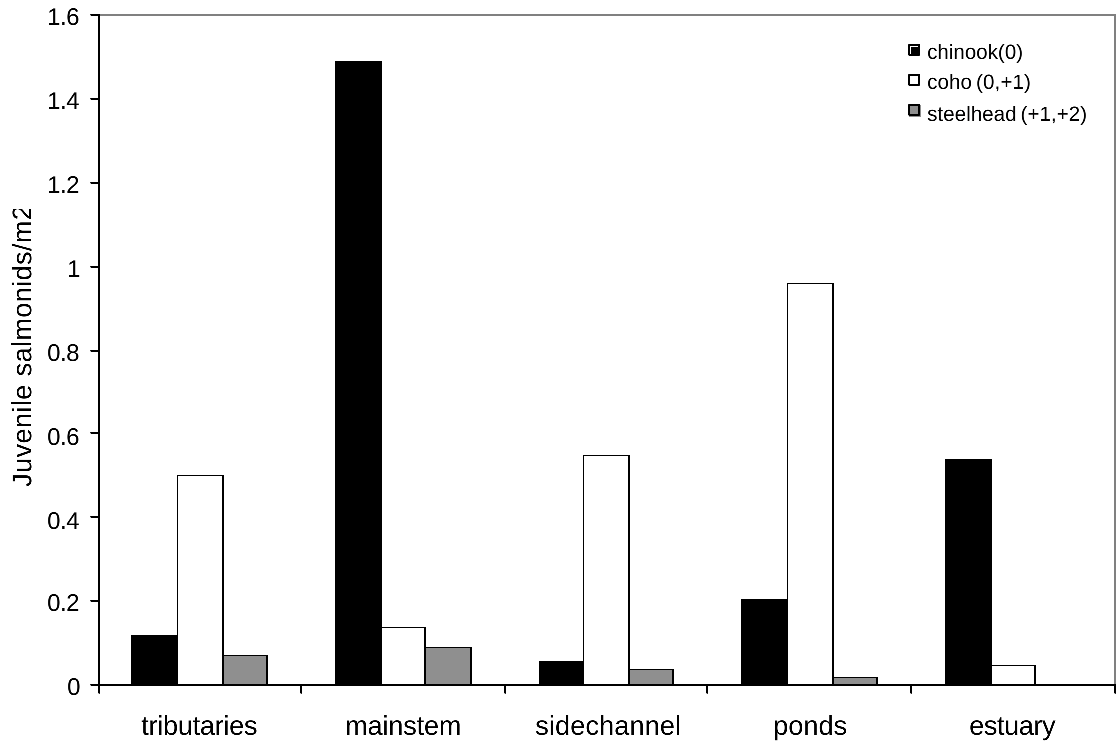

Figure 4-5. General juvenile salmonid use at the habitat scale. Compilation of over 60 references.

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Additional references to add

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CHAPTER 5. USING RESULTS OF ASSESSMENTS TO PRIORITIZE SPECIFIC RESTORATION ACTIONS

Phil Roni, Timothy J. Beechie, and George R. Pess

Previous chapters have outlined methods for assessing habitat loss and degradation and estimating fish response to changes in habitat. The methodologies outlined in these chapters help identify disrupted processes and restoration opportunities. For example, a culvert inventory would provide a list of opportunities for culvert replacement to improve fish access, or an inventory of riparian areas might identify opportunities for replanting, thinning, or fencing (Table 5-1). The next step is to prioritize specific restoration actions within a watershed that were identified as restoration opportunities during the assessments. However, this requires an understanding of the watershed process or function that specific technique is likely to restore, the effectiveness and likelihood of success of each technique, and the potential fish response to that technique. Numerous techniques exist to restore disrupted processes or enhance habitat in the short term. For example, several types of wood placement have been developed to increase instream habitat complexity and loss of large woody debris (LWD). Roni et al. (in press) recently reviewed these various techniques. In this chapter we describe the effectiveness of various restoration techniques and a method for prioritizing restoration actions within a watershed.

Background—What Do We Know About Restoration?

The term "restoration" has been used to describe a suite of habitat manipulations, enhancements, and improvements. In its strictest definition, restoration is returning a site to some predisturbance condition. However, here we will use the term generically to mean efforts to improve or enhance watershed and habitat conditions. Restoration can also be further classified as passive and active. Passive techniques seek to restore processes and set up conditions that will allow recovery of the stream (e.g., exclusion of cattle from riparian areas, replanting riparian vegetation, increasing instream flows) or system. Active techniques are those that seek to directly manipulate habitat, such as removal of migration barrier or placement of logs in a stream channel to create pools (find a formal definition of these and insert). These tend to create rapid changes in habitat and treat symptoms of disrupted watershed processes rather than restore the process.

Watershed and stream restoration are a key component of many land management plans, and should be an important component of most recovery plans for threatened and endangered salmon. Tens of millions of dollars are spent annually in individual river basins in an effort to enhance or restore habitat for salmonids and other fish species (NRC 1996). The majority of these dollars are being allocated to local citizen watershed groups for watershed restoration and recovery. Unfortunately, local citizen groups often lack adequate guidance on which types of restoration or enhancement to conduct first or which techniques are most successful. More importantly, it is often unclear how individual site-specific actions might fit into a larger context of watershed restoration and recovery of salmon stocks.

In part, the lack of guidance stems from limited information on the biological effectiveness of various habitat restoration and enhancement techniques and the need for comprehensive evaluation and monitoring (Reeves et al. 1991, Frissell and Nawa 1992, Chapman 1996). The response of fishes to watershed and stream habitat restoration techniques (e.g., instream structure placement, riparian planting, road restoration, and reconnection of isolated habitats) have not been thoroughly evaluated, and there is considerable debate within the scientific community about the effectiveness of various techniques (Reeves et al. 1991, Kondolf 1995, Kauffman et al. 1997). Most monitoring has focused on the physical response to various instream restoration techniques with inadequate monitoring of fish, invertebrates, and other biota. Response of fish and other biota are inherently more difficult to monitor than physical conditions. However, the biological response to various restoration techniques is the ultimate measure of restoration effectiveness. The large interannual variability of juvenile and adult salmonid abundance often requires more than 10 years of monitoring to detect a response to restoration (Bisson et al. 1992, Reeves et al. 1997). Existing monitoring has also indicated highly variable results from some techniques such as wood and boulder placement in streams (Chapman 1996). Therefore, drawing conclusions about the biological effectiveness of various techniques has been difficult and has hampered efforts to provide scientific guidance on restoration activities.

However, we do have a reasonable understanding of the processes that affect channel morphology and create fish habitat (see previous chapters). In the coastal Pacific Northwest, for example, the delivery of organic matter (e.g., woody debris and leaf litter), water, and sediment are some of the major processes dictating channel morphology and the formation of habitat (Montgomery and Buffington 1998). In the 1990s it became widely accepted that restoring watershed processes is the key to restoring watershed health and improving fish habitat. Beechie et al. (1996), Kauffman et al. (1997), Beechie and Bolton (1999), and others have described restoration and recovery strategies that place emphasis on restoring physical and biological processes that create healthy watersheds and high-quality habitats. Activities that restore processes (e.g., road removal and restoration, culvert removal, and riparian and upslope restoration) are often conducted at the site or reach level. A method is needed that places site-specific restoration within a watershed recovery planning efforts. In this chapter we provide a hierarchical strategy for prioritizing site-specific restoration activities within a watershed.

As identified in previous chapters, watershed assessments are a critical first step to understanding watershed processes and identifying restoration needs within a watershed for recovery planning. However, before one can prioritize specific restoration actions within a watershed, a thorough understanding of the physical and biological effectiveness of various restoration methods is also needed. Roni et al. (in press) recently reviewed common restoration techniques, their effectiveness, longevity and whether they restore processes or are short term habitat enhancement (Table 5-2). Most restoration techniques fall into five general categories: 1) habitat reconnection, 2) road improvement, 3) riparian restoration, 4) instream habitat restoration, and 5) nutrient enrichment. Within these five general categories, several other specific techniques can be identified. The information from Roni et al. (in press) was used as a basis for prioritizing restoration techniques and identifying additional research and monitoring needs.

Strategy for Prioritizing Actions

The prioritization of restoration actions may be based on a number of factors, including the needs of individual species, locations of refugia, or cost-effectiveness (Beechie and Bolton 1999). It is also important to consider the response time, probability and variability of success, and the duration of a given restoration action (Table 5-2). Those techniques that have a high probability of success, low variability among projects, and relatively quick response time should be implemented before other techniques. For example, reconnecting isolated off-channel habitats or blocked tributaries provides a quick biological response, is likely to last many decades and, based on available evidence, has a high likelihood of success. Generally, these types of restoration activities should be undertaken before methods that produce less consistent results. Riparian restoration or road improvement may not produce results for many years or even decades for some functions (Table 5-2) and should be considered after reconnecting high-quality isolated habitats. Other techniques, such as instream LWD placement or other instream restoration, are generally effective at increasing coho salmon densities. However, instream actions such as these are habitat manipulations or enhancements and should be undertaken either after or in conjunction with reconnection of isolated habitats and other efforts to restore watershed processes. In addition, manipulation of instream habitat may be appropriate where short-term increases in fish production are needed for a threatened or endangered species (Beechie and Bolton 1999).

Roni et al. (in press) developed a hierarchical flow chart that can be used to help guide the selection and prioritization of restoration projects based on these principles (Figure 5-1). This flow chart combines the known effectiveness of various techniques with the need to restore habitat-forming processes (Figure 4-1) identified by watershed assessments (Chapter 4) and the protection of high-quality habitats. Ideally, habitat restoration requires reconnecting isolated habitats and restoring the disrupted habitat-forming processes (Beechie and Bolton 1999). Habitat manipulations (i.e., instream structures) are generally unnecessary except where adjacent land uses constrain restoration options. In such areas, instream projects that are consistent with the natural habitat characteristics of the site are an option.

While most techniques fit well into this hierarchy, carcass placement and nutrient enhancement and estuarine restoration are new techniques whose place in it is uncertain. Little is known about the effectiveness of estuarine restoration. However, reconnecting isolated estuarine habitats such as distributary sloughs is similar to reconnecting isolated off-channel habitats, which has been shown to be effective (Table 5-2). Furthermore, given the importance of estuaries to anadromous fishes and the success of reconnecting isolated off-channel habitats, it is likely that reconnecting estuarine habitat would be effective and should be considered at the same time as reconnecting other isolated habitats. The placement of salmon carcasses or other nutrients into streams may increase fish condition and production in the short term. This restoration technique is a form of habitat enhancement that can occur at any stage in the watershed restoration process. However, because it does not restore but rather mitigates for a deficient process, we have suggested that it be considered at the same point in the hierarchy as instream habitat manipulation. Similarly, the creation of new estuarine or off-channel habitats does not restore a process and the effectiveness of these efforts is unclear.

A common restoration technique not covered in Roni et al. (in press) is restoration of instream flows or natural hydrology either from water withdrawal projects or below large water storage projects. Water withdrawal or flow manipulation disrupts hydrologic processes, including delivery and routing of sediment and nutrients, and can dramatically impact habitat formation, connectivity, and quality (Bednarek 2001). We consider restoring instream flows and natural hydrologic patterns part of reconnecting isolated habitats, and therefore do not have a separate category for this technique.

Within the broad restoration categories in Figure 5-1, some techniques are more effective than others or more applicable in some provinces than others. For example, we include riparian silviculture with fencing and reduced grazing under riparian restoration. Livestock exclusion is a form of riparian protection that has been shown to be effective on range and agricultural lands (Platts 1991), while the long-term effectiveness of riparian replanting and conversion techniques is largely unknown. Priorities for different types of riparian restoration will differ by region and watershed as will other specific restoration techniques that fall into the broad categories we have defined. However, a watershed assessment is the important first step to determine the most effective type of restoration within a given restoration category for the watershed in question.

We also separate the placement of instream LWD or boulders into reaches with high production potential and low production potential. Low-gradient channels (<5% slope) are the stream reaches most frequently used by Pacific salmon (Montgomery et al. 1999) and also where LWD additions are known to provide physical and biological benefits. Therefore, in the cases where instream restoration techniques are implemented, they should occur in reaches with gradients less than 5%. Placing wood or other structures in steeper channels is less likely to have the desired physical or biological benefits.

Our approach was primarily designed for forest, range, and other moderately modified rural lands. In urban areas, hydrologic and sediment processes in streams are highly altered (e.g., increased high flows and channel down-cutting). Areas with intensive agriculture often have severe water quality problems, and stream channels in both urban and agricultural areas are often highly channelized and lack adequate riparian vegetation. A combination of urban and agricultural impacts may also inhibit restoration of estuarine habitats. The principles outlined in above and in Figure 5-1 are still useful in urban and agricultural lands, but in these environments other factors such as large infrastructure (e.g., highways and buildings) may constrain restoration opportunities. Thus the framework we outline may need to be modified for use in these highly altered systems where some processes cannot be reliably restored, or where water quality or hydrologic changes may compromise the effectiveness of many of the commonly employed restoration techniques. A more detailed watershed assessment and restoration prioritization technique such as that outlined by the Skagit Watershed Council (1999) may be useful in these areas.

Need for Monitoring and Management Experiments

Our review of various restoration techniques indicates that knowledge about the effectiveness of most techniques is incomplete and comprehensive research and monitoring are needed. Even techniques that appear to be well studied, such as instream LWD placement, need more thorough evaluation and long-term monitoring. This emphasizes the need for comprehensive monitoring and evaluation of both individual and multiple restoration actions at multiple scales. Many restoration actions should be treated as management experiments and accompanied by research and monitoring to determine both physical and biological responses. These results can then be used to guide future restoration actions and more accurately quantify the potential increase in fish production for habitat manipulations.

Finally, we view the method outlined in this chapter as the first iteration of guidelines for prioritizing site-specific restoration activities and for providing guidance for salmon recovery planning. Our approach is designed for watersheds where detailed information on processes and stream reaches is not initially available. However, we view watershed assessment and restoration as an iterative process. As more information becomes available on a specific watershed and the effectiveness of various techniques for salmonids and other fishes, our approach should be modified.

Summary

The methodology we present for prioritizing site-specific restoration strategies in a watershed context (Figure 5-1) is drawn from Roni et al. (in press) and based on three key elements: 1) principles of watershed processes (i.e., Figure 4-1), 2) protection of existing high-quality habitats, and 3) current knowledge of effectiveness of specific techniques (Table 5-1). We view techniques that manipulate instream habitat as the final step in the hierarchical strategy because they tend to be short lived, the results are highly variable among techniques and species, and they do not seek to restore processes (Table 5-2). While we focus on restoration techniques in this chapter, it is important not to overlook the need to protect high-quality habitats. Protection of high-quality habitat should be given priority over habitat restoration, as it is far easier and more successful to maintain good habitat than to try and recreate or restore degraded habitat. Furthermore, our recommendations are dependent upon a watershed assessment (Chapter 4) and in no way negate the need for adequate assessment of processes and current conditions in a watershed.

Table 5-1. Examples of assessment methods and possible types of restoration actions that may be identified.

Assessment method	**Examples of impaired processes identified or information obtained**	**Example of possible types of restoration actions identified**
Road and culvert inventory	Sediment sources, landslide hazard, habitat connectivity, hydrology, nutrient transport	Culvert removal or replacement, road restoration or removal
Riparian inventory and assessments	LWD and organic matter delivery, shade,	Replanting, conifer conversion, fencing projects
In-channel habitat survey	LWD delivery, habitat complexity	Instream restoration (LWD, nutrient enhancement etc.
Estuarine and floodplain connectivity	Habitat connectivity, LWD delivery	Remove dikes, tide gates, reconnect isolated habitats

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Table 5-2. Typical response time, duration, variability in success and probability of success of common restoration techniques (modified from Roni et al. in press).

Restoration type	Specific action	Years to achieve response	Longevity of action (y)	Variability of success among projects	Probability of success
Reconnect habitats	Culverts	1-5	10-50+	Low	High
	Off channel	1-5	10-50+	Low	High
	Estuarine	5-20	10-50+	Moderate	Moderate to high
Road improvement	Removal	5-20	Decades to centuries	Low	High
	Alteration	5-20	Decades to centuries	Moderate	Moderate to high
Riparian vegetation	Fencing	5-20	10-50+	Low	Moderate to high
	Riparian replanting	5-20	10-50+	Low	Moderate to high
	Rest-rotation or grazing strategy	5-20	10-50+	Moderate	Moderate
	Conifer conversion	10-100	Centuries	High	Low to moderate
Instream habitat restoration	Artificial log structures	1-5	5-20	High	Low to high[a]
	Natural LWD placement	1-5	5-20	High	Low to high[a]
	Artificial log jams	1-5	10-50+	Moderate	Low to high[a]
	Boulder placement	1-5	5-20	Moderate	Low to high[a]
	Gabions	1-5	10	Moderate	Low to high[a]
Nutrient enhancement	Carcass placement	1-5	Unknown	Low	Moderate to high
	Stream fertilization	1-5	Unknown	Moderate	Moderate to high
Habitat creation	Off channel	1-5	10-50+	High	Moderate
	Estuarine	5-10	10-50+	High	Low
	Instream	See various instream restoration techniques above			

[a] Depends upon species and project design

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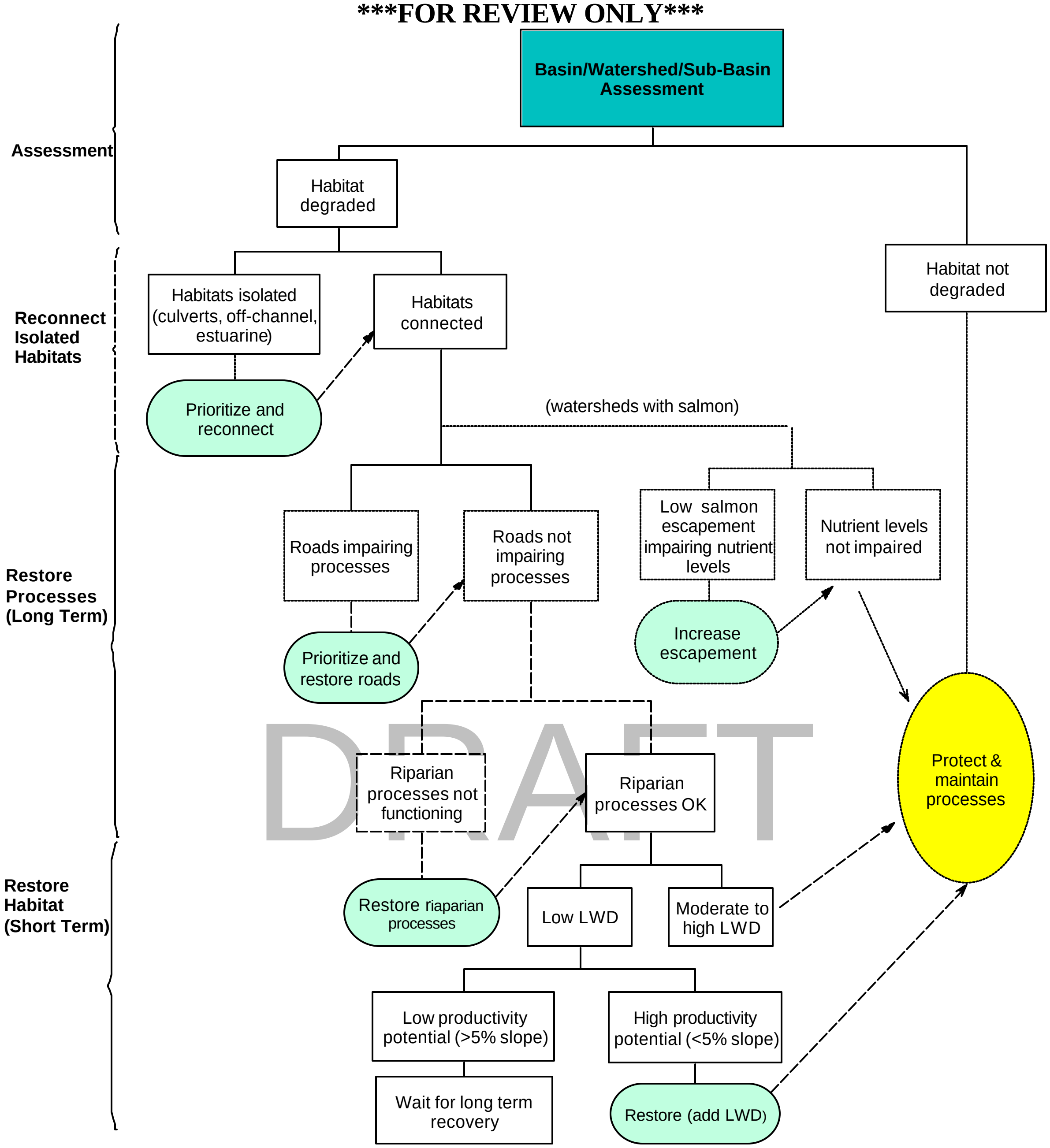

Figure 5-1. Flow chart depicting hierarchical strategy for prioritizing specific restoration activities (modified from Roni et al. in press). Shaded boxes indicate where restoration actions should take place. Addition of salmon carcasses or nutrients may be appropriate at various stages following reconnection of isolated habitats.

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Citations – Ch. 5

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CHAPTER 6. UPDATING THE RECOVERY PLAN

[NOTE: This chapter is awaiting content.]

CHAPTER 7. ISSUES OF SCALE IN HABITAT RECOVERY PLANNING

Cara Campbell

The aquatic environment is complex and dynamic, changing continually across space and time. Inhabitants of this environment have evolved to these ever-changing conditions. However, anthropogenic alterations to the landscape have disrupted the natural processes within these systems and species are forced to contend with excessive, unnatural conditions. These alterations can be large or small, influencing expansive areas or more local conditions, and the effects can occur immediately or years later. Thus a thorough knowledge of the processes structuring the aquatic environment and how these processes interact over various spatial and temporal scales is inherent to understanding the effects of disturbance on aquatic systems and their inhabitants. This chapter will discuss the concept of scale and how it can be incorporated into recovery planning.

Hierarchical Nature of Stream Systems

Stream systems can be thought of hierarchically (Frissell et al. 1986), organized into nested levels. Processes occurring at each level in the hierarchy tend to occur at similar frequencies and correspond spatially (Urban et al. 1987), as well as generate conditions that influence the environment at the next level (Frissell et al. 1986, Urban et al. 1987). For example, Rot et al. (2000) described this hierarchical interrelationship of processes using valley constraint, riparian landform, riparian plant community, channel type, and channel configuration. At a large scale, valley constraint influences riparian landform development. These riparian landforms subsequently control the impact of fluvial disturbances on the composition of the riparian plant community. Successional processes within the riparian forest then influence the size of LWD within the stream channel. Finally, the resultant channel type and configuration reflect this pathway of processes as well as local conditions. Thus, the physical and biological conditions found at a particular location within the stream system are products of processes occurring at various spatial and temporal scales. These various processes result in a continuous gradient of physical and biological conditions along a stream system that determine subsequent community structure and function (Vannote et al. 1980). Thus, a thorough understanding of the linkages between these biological and physical processes within and across scales is required for effective management of habitat and defining recovery potential (Lewis et al. 1996).

Scale

Scale can be broadly defined as denoting “resolution within the range of a measured quantity” (Schneider 1994). Three aspects of scale commonly used in investigating aquatic systems are space, time, and resolution (Figure 7-1). Space takes into account a physical region or area, time incorporates temporal extent, while the degree of resolution identifies ecological organization (e.g., taxonomic or functional groupings) (Frost et al. 1988). Definitions of the various terms used on the space and resolution axes can be found in Table 7-1.

Scale is an important consideration in any study, for it dictates the methods used, results obtained, and any subsequent interpretations (Wiens 1986), and reflects the different questions, objectives, and goals of the investigators (Hinch 1991). Depending on where your examinations focus along each axis in Figure 7-1 (i.e., what scale), different patterns and explanatory mechanisms can emerge (Wiens et al. 1986, Frost et al. 1988, Hinch 1991; Levin 1992). Generally, larger scale work tends to focus on identifying patterns and possible processes controlling these patterns, while smaller scale examinations concentrate on elucidating the mechanisms underlying these processes (Urban et al. 1987, Levin 1992). Thus, key information can be obtained at any scale of study; however, combining this knowledge across scales and disciplines has proven inherently difficult (Imhof et al. 1996). That is problematic, for just this combination of knowledge is required to improve our understanding of the behavior of the complex aquatic environment (Urban et al. 1987, Levin 1992). The next sections will illustrate the types of information sought in small- and large-scale studies and attempts to combine this information in multi-scales studies.

Small-Scale Studies

Much of what is known about species-habitat relationships and biotic interactions (competition and predation) comes from smaller scale studies. Species-habitat relationships have uncovered local physical features used by individuals, how this habitat use changes ontogenetically and temporally, and how these habitat preferences differ by species. For example, juvenile coho (*Oncorhynchus kisutch*) are more closely associated with pool habitat, while steelhead (*Oncorhynchus mykiss*) inhabit a wider array of habitats (Bisson et al. 1988, Fausch 1993). Also, the amount of both suspended and deposited sediment can adversely affect egg and juvenile salmonid survival. These types of studies have also identified bottlenecks limiting the production of different salmonids. For example, the amount of LWD contributes to the proportion of pools and has been linked to salmonid survival, particularly serving as velocity refuges during the winter (Bustard and Narver 1975, Tschaplinski and Hartman 1983).

Subsequently, the availability of suitable winter habitat can limit coho production (Solazzi et al. 2000). Understanding these species-habitat requirements is essential for successful restoration and conservation efforts because species with rigid habitat requirements are under a stronger threat from degraded habitat conditions than those with flexible habitat requirements.

Smaller scale studies have also contributed to our understanding of both intra- and inter-specific competition. These interactions can cause habitat shifts, reducing the fitness of displaced individuals. For instance, 0+ Atlantic salmon (*Salmo salar*) generally inhabit shallow riffles with coarse substrate, moving into deeper riffles with pools as they grow, at which time they can either be viewed as prey or chased back into shallow habitat by the larger 1+ individuals (Symons and Heland 1978). Similarly, Hearn and Kynard (1986) showed interspecific interactions to result in habitat segregation among 1+ Atlantic salmon and rainbow trout. The degree of similarity in habitat requirements between species as well as the similarity in size of individuals both within and across species influences the degree of competition for space within a particular reach or habitat unit, determining the likelihood for biotic interactions. Also, a species' degree of tolerance to degraded conditions can determine the likelihood of its displacement by another species (Nelson et al. 1992). Thus, any alterations affecting the quantity or quality of the habitat can shift the balance in favor of one species or life stage over another.

The above examples were not presented to illustrate all factors influencing salmonids within stream systems or all types of studies currently being conducted, but rather to convey the type of information generated from these small-scale endeavors. Through these small-scale studies, much of our knowledge about both biotic interactions and associations between salmonids and the local stream environment is known, and subsequently, about the biological and physical consequences of processes occurring over various spatial and temporal scales. These controlled, manipulative investigations tend to utilize basic field sampling methodologies such as habitat surveys, snorkeling, and electrofishing. The studies have amassed large quantities of data covering small spatial scales over short time frames, and the species-habitat relationships uncovered are transferable across systems. However, both inherent and anthropogenic differences across watersheds (e.g., geology, landform, land use, climate, etc.) combine to produce a different suite of conditions within each watershed. For example, shallow streams are more variable, exposing the associated communities to a greater range of conditions (Jackson et al. 2001), thus systems composed of a larger number of shallow streams will present different challenges to its inhabitants than a watershed containing a larger proportion of deeper streams. The combination of varying conditions and the stochastic disturbances common over smaller scales results in these studies being more variable, thus, less predictable and less repeatable (Levin 1992). As a result, there is a growing trend toward examining larger spatial scales over longer time frames, giving up some of the detail of the small-scale studies for the sake of more generalized behaviors, and subsequently, greater predictability (Levin 1992).

Large-Scale Studies

Large-scale investigations generally address questions regarding climate change, geomorphic processes, and anthropogenic disturbance, and the role of each in shaping community structure. Climate change is often studied via examinations of temperature regimes, changes in these temperature regimes over time, and their role in determining species distributions. In general, water temperature can structure fish assemblages (Waite and Carpenter 2000) as well as species distributions (Dunham et al. 1999). Increases in water temperatures associated with global warming can potentially reduce suitable salmonid habitat through an increase in both maximum summer temperatures and minimum winter temperatures, potentially restricting cool water species to higher elevations and increasing the likelihood of localized extinctions as well as both habitat and population fragmentation (Keleher and Rahel 1996). Also, enhanced embryonic development during the winter and spring can alter the timing of fry emergence, having both positive and negative effects depending on the species and its life-history strategy (Hartman et al. 1996). Thus, understanding the extent and timing of changes in temperature regimes in the context of life-history strategies and habitat requirements can uncover potential losses of critical habitat and subsequent alterations of population and community structure.

Much of our knowledge of the role of geomorphic variables in influencing species distribution and abundance arise from larger scale studies. Within river systems, physical characteristics of streams, such as discharge, channel geometry, channel pattern, and size of tributaries exhibit regular longitudinal patterns (Allan 1995). These regular patterns allow for the generalization of habitat characteristics within larger geomorphic variables. For example, Nelson et al. (1992) found sedimentary streams to be narrower, with coarser substrate and lower embeddedness than volcanic or detrital streams. Rot et al. (2000) found bedrock channels to be in constrained valleys, with low volumes of LWD, and limited alluvial landform development. In contrast, forced pool-riffle channels were located in moderately constrained to unconstrained valleys, with higher volumes of LWD, and extensive alluvial landforms (Rot et al. 2000). Once established, these associations allow for the identification of relationships between community or population structure and these larger physical characteristics. For example, Lanka et al. (1987) found a reduction in trout density with increasing stream size, possibly as a result of decreased riparian cover or an increase in human disturbance, while Platts (1979) found chinook (*Oncorhynchus tshawytscha*) and steelhead spawning and rearing areas concentrated within third through fifth order streams. Bull trout (*Salvelinus confluentus*) have been associated with alluviated lowland and valley streams (Watson and Hillman 1997), while steelhead have shown strong negative correlations to areas comprised of greater than 10% unconsolidated lithology (Thompson and Lee 2000). Thus, associations between larger scale geomorphic variables and more detailed underlying habitat characteristics enables identification of patterns between species and these geomorphic variables in areas without specific habitat measurements.

Land use and its impacts on aquatic species have also been investigated by large-scale studies. For example, logging and road construction can reduce the amount of LWD and increase sediment supply, eliminating off-channel habitats. Dams can block migration pathways, fragmenting spawning and rearing habitat (Dauble and Geist 2000) as well as populations. Agricultural lands can be associated with nutrient loading, pesticide inputs, and reduced riparian vegetation. Urbanization can be associated with increased pollution and further road production. Since greater habitat complexity is associated with greater community complexity (Gorman and Karr 1978) and anthropogenic actions generally serve to reduce complexity, understanding how the various land use activities alter natural processes and conditions within stream systems is critical for recovery efforts.

Again, these examples are not meant to provide an exhaustive list of the large-scale research being conducted, but rather to provide a glimpse of the information sought from these larger scale investigations. Generally, these studies seek to identify patterns or processes occurring across systems. They incorporate the use of alternative methodologies such as aerial photography, GIS, and spatial statistics and modeling, as well as the aggregation of smaller habitat units or reaches to examine larger areas. These methodologies provide a way of examining and comparing systems quickly, without conducting actual field investigations; however, they can only elucidate patterns, not causation. As a result, it is necessary to utilize the smaller scale species-habitat studies to identify the underlying mechanisms that could be responsible for these larger scale patterns. Thus, combining the use of small- and large-scale studies provides the most comprehensive view of aquatic systems, as is illustrated by the increase in multi-scale studies found in the literature.

Multi-Scale Studies

Multi-scale investigations seek to understand the larger scale variables structuring aquatic species and, within these larger variables, the specific characteristic influencing abundance, distribution, growth, etc. For example, an examination in the Willamette River basin in Oregon found water temperature and stream slope to structure fish assemblages across ecoregions (large-scale), while water chemistry was the dominant influence within ecoregions (Waite and Carpenter 2000). Dunham and Rieman (1999) found bull trout to occupy patches (defined as watersheds >400 ha) with the greatest area and the lowest road density, while within these occupied patches, bull trout were associated with wider streams. Valley bottom type has also been implicated as a key component determining bull trout presence at both site and smaller, habitat scales (Watson and Hillman 1997). It is also possible for environmental processes to influence the different components of production according to scale. For example, larger scale temperature gradients can influence the growth and size of juvenile Atlantic salmon within a watershed, while small-scale temperature profiles inhibit survival within subwatersheds (Campbell 1999). Similarly, Pyper et al. (1999) found between-region processes to influence length of sockeye (*Oncorhynchus nerka*) within the marine environment, while survival rate was related to within-region processes. As species assemblages and distributions are structured by a combination of larger scale geomorphic and climatic conditions and the more specific biotic and abiotic conditions of the local environment, it is necessary to conduct studies at an array of spatial and temporal scales. These multi-scale studies can identify the way larger scale variables dictate potential fish assemblages and species ranges and how these are realized within the constraints of the local environment.

Scale in Recovery Planning

Different questions are addressed at different scales of inquiry and much information about aquatic systems and their inhabitants is gained at each. Therefore, successful restoration of both natural processes and salmon populations requires the incorporation of this multi-scale information into the recovery planning process. The first step is inventorying and collecting all available data, covering multiple temporal and spatial scales, across the area of consideration. This would initially focus on current and historical data of three types: 1) status of habitat forming processes and biological integrity, 2) condition and distribution of aquatic habitats, and 3) abundance and distribution of salmon (Chapter 4). This information then needs to be examined to assess how habitat factors prevent populations or their parent ESUs from meeting the four types of recovery goals: abundance, productivity, diversity, and spatial structure (McElhany et al. 2000). The following sections will discuss how scale can be incorporated into assessing how habitat changes might have altered the abundance, productivity, diversity, and spatial structure of populations, and subsequently ESUs.

Abundance

The risks to a population are inversely related with abundance, so it should be possible to define broad risk categories based on abundance (McElhany et al. 2000). In this sense, abundance categorization can be viewed as a large-scale question. To address how habitat changes might have altered abundance, it is necessary to determine how abundance of species changes with land use. First, it is important to identify potential species distributions over the area of interest, incorporating broad habitat characteristics as well as information obtained from the literature. For example, Nelson et al. (1992) found Lahontan trout (*Oncorhynchus clarki henshawi*) presence primarily within the sedimentary geologic district of the North Fork Humboldt River drainage, while Dunham et al. (1999) identified the downstream distribution limit as elevations with an average July air temperature of 18ºC. Next, associating abundance estimates with broad characterizations of anthropogenic activity can illustrate where land-use alterations have constrained suitable habitat, thereby reducing abundance. For example, Bradford and Irvine (2000) used spawner abundance data within the Thompson River, British Columbia, to examine relationships between land use within a watershed and changes in recruitment to individual streams. They found that coho populations declined in relation to the extent of agricultural and urban land use as well as the density of roads within the watershed. Paulsen and Fisher (2001) estimated juvenile spring-summer chinook parr-to-smolt survival within the Snake River ESU utilizing PIT tag information and related this back to broad characterizations of land use. They found that wilderness areas and areas with decreased road density are associated with higher overwinter, subsequently parr-smolt, survival. If smolt-to-adult survival estimates could be linked with this type of data, it might be possible to make broad abundance predictions based on land use. These can be coarse investigations that identify where land use activities might be reducing the abundances of populations and decreasing their viability or that of their parent ESU's. This can be a useful first step that can focus subsequent, more detailed investigations of the actual processes being impaired and the recovery efforts needed.

Productivity

The productivity of a population can be viewed in terms of sustained trends in abundance, which can be assessed by examining population growth rate: production realized over the entire life cycle (McElhany et al. 2000). In examining how habitat changes might have altered productivity of a population, it could be useful to concentrate on investigating stage-specific carrying capacity. This would require a smaller scale approach, highlighting, for example, the smolt production sustainable by the environment and any habitat characteristics that could be limiting this realized carrying capacity. For example, Nickelson et al. (1992) used this approach to estimate coho smolt production for coastal Oregon basins using juvenile density estimates by habitat type for different seasons. Fully-seeded streams were sampled each season and habitat was classified using a modified version of the habitat classification scheme described by Bisson et al. (1982). A total of 15 streams and 150 habitat units were sampled and density estimates were generated for each habitat type. Juvenile coho were most abundant in pool habitat throughout the year, but the type of pool utilized varied with season. During spring, fry were predominantly in backwater pools, while in the winter parr densities were highest in alcoves and dammed pools (the majority of which were beaver ponds). As alcoves and beaver ponds composed only 9% of the total number of habitat units sampled and 31% of the winter area sampled, they concluded that production of smolts in Oregon coastal streams can be limited by the availability of winter habitat. Similar types of analyses can be conducted over streams and their results extrapolated to watersheds or ESUs, scales at which a large percentage of the habitat remains unsurveyed. They can also be repeated for various life stages and species, identifying potential overlap of critical habitat requirements. Once these bottlenecks are identified, they can aid in the recovery planning process by measuring the quantity of this critical habitat available in non-impacted sites versus impacted sites. Also, historical reconstructions can be conducted to quantify the actual losses of these critical habitats in impacted areas. This information is useful in the recovery process, for it can highlight the processes that need to be restored to improve the health of the local habitat as well as set recovery priorities based on the degree of degradation. See Appendix B for a more detailed examination of this process.

Diversity

Diversity refers to the distribution of traits within and among populations (McElhany et al. 2000). These traits can exhibit considerable variation within and among populations and can result as a combination of genetic and environmental factors (McElhany et al. 2000). The genetic controls are outside of the scope of this document, but it is possible to examine how habitat changes might have altered diversity by larger scale examinations of traits such as size of returning adults and age or size of smolts. Smolt production is influenced by smaller scale, watershed-specific landscape (<30 km) and biotic factors (Bradford 1999). Thus, using control and treatment streams with similar underlying habitat structure (hydrology, geomorphology, ecology) (Bradford 1999) it may be possible to investigate how different forms or degrees of land use have impacted the age or size of smolts. For example, higher temperatures are known to increase growth rate (Beckman et al. 1998) that is in turn associated with reduced smolt age

(Beckman et al. 1998, Connor et al. 2001). Logging, agriculture, and urbanization all can potentially reduce riparian vegetation, decreasing shading and increasing temperatures within the stream channel, so it should be possible to study the size and age of smolts leaving control streams with minimal land use versus treatment streams with varying degrees of degradation. It could also be useful to couple habitat productivity indices with smolt age information. For example, Connor et al. (2001) found Snake River wild subyearling spring-summer chinook that migrate to the Snake River main stem grow faster, migrating as subyearlings, while those that remain in natal streams migrate as yearlings. They concluded that the main stem provides better growth opportunity than the less productive natal streams. It can also be useful to examine size at return to a specific system over time. Adult female size has been linked to freshwater migration distance (Fleming and Gross 1989) and competition on spawning grounds (Van Den Berghe and Gross 1989). Egg size is generally correlated with body size and the advantages of each can vary with the quality of the spawning site (Van Den Berghe and Gross 1989). Thus, changes to the landscape that alter the quality of spawning grounds can shift the size structure of spawners and affect the size, quality, and survival of eggs and juveniles.

Spatial Structure

The spatial structure of a population refers to the spatial distribution of individuals within a population and the processes generating that distribution, and is dependent on habitat quality, spatial configuration, and the dynamics and dispersal characteristics of individuals in the population (McElhany et al. 2000). As the stream environment is heterogeneous, it may be viewed as a series of “habitat patches” at an array of spatial scales. As habitat quality varies across patches, the likelihood of individuals inhabiting each patch is dependent on the quality of the habitat in the patch as well as the ability of individuals to move between patches. Therefore, a proper investigation into how habitat changes might have altered the spatial structure of a population requires understanding the small- and large-scale influences on both habitat patch dynamics and salmonid movement. For example, in an examination of Mexican spotted owl (*Strix occidentalis lucida*) habitat, Keitt et al. (1997) examined the importance of habitat patches at several maximum inter-patch dispersal distances. At dispersal distances between 0 and 40 km, patch size was the strongest indication of the patches importance to connectivity, while above 60 km, the number of alternative pathways increased to the degree that removing patches had little effect on connectivity. It was dispersal distances between 45 and 50 km that identified "stepping stones" or critical patches connecting large habitat areas. As a result, they concluded that the species whose dispersal abilities are within this latter range will be dependent on landscape configuration, for individual patches can act as corridors, bridging areas of suitable habitat. Similarly, Dunham and Rieman (1999) examined 104 patches (defined as catchments draining >400 ha) within the Boise River to identify patterns in juvenile bull trout occurrence. They utilized digital road and hydrography coverages to assess road densities by patch and inter-patch distances, respectively. Bull trout occurrence was positively related to patch size (area) and negatively related to both inter-patch distance and road density. Thus, Dunham and Rieman (1999) concluded that populations in larger, less isolated, and less disturbed habitats are more likely to persist, while small, isolate, disturbed populations are at risk. Identifying how species utilize habitat patches in association with general knowledge of their dispersal characteristics is a necessary first step in the recovery planning process. Once identification of critical habitat patches is achieved, it is possible to examine how land use actions have reduced or eliminated connectivity of these patches as well as altered or destroyed the patches themselves. For example, Beechie et al. (1994) identified hydromodification as the greatest cause of lost smolt production within the Skagit River, followed by the isolation of 37 km of tributaries above 33 blocking culverts. This information can then dictate the scale of recovery efforts by pinpointing the processes that are in need of restoration or identifying local, specific recovery actions.

Conclusion

Scale is an important consideration in the recovery planning processes. Certain questions can be easily addressed at different scales (e.g., abundance can be categorized in relation to large-scale natural processes over broad areas and long time frames). However, these specific aspects of population viability are not mutually exclusive and the information gleaned at one scale can easily be incorporated into addressing a separate aspect at a different scale. For example, it would be feasible to incorporate the large-scale distribution information and the smaller scale estimates of carrying capacity together to aid in identifying patches and their connectivity. The result is multi-scale analyses into the various components of population and ESU viability, the hierarchy of natural processes determining this viability, as well as the recovery actions needed to restore these natural processes.

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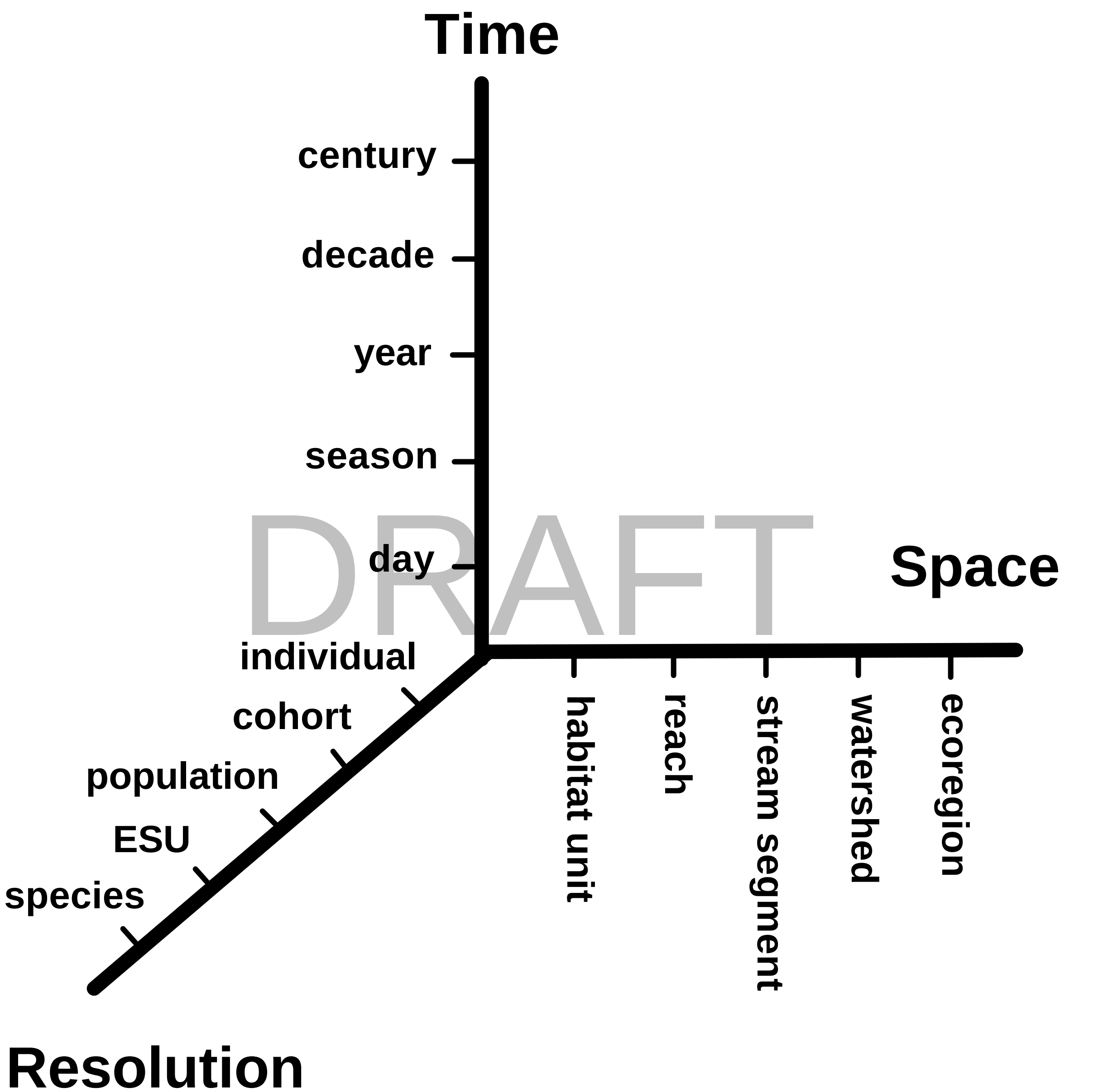
Figure 7-1. Three aspects of scale commonly used in investigating aquatic systems (adapted from Frost et al. 1988). Factors on the interior portion of the plot are addressed at small scales and those found on the outside are the focus of larger scale studies.



*****FOR REVIEW ONLY*****

Table 7-1. Definitions for the terms found on the space and resolution axes in Figure 7-1.

Term	Definition
Space:	
Habitat unit	Relatively homogenous area of the stream channel that differs from adjoining areas in depth, velocity, and substrate characteristics (Armantrout 1998)
Reach	Section of a stream segment between named tributaries, changes in valley and channel form, major changes in vegetation type, or changes in land use or ownership (Moore et al. 1997)
Stream segment	Portion of a stream system flowing through a single bedrock type and bounded by tributary junctions or major waterfalls (Frissel et al. 1986)
Watershed	Region or area drained by surface and groundwater flow in rivers, streams, or other surface channels (Armantrout 1998)
Ecoregion	An area determined by similar land surface form, potential natural vegetation, land use, and soil; may contain few or many geological districts (Omernik 1986)
Resolution:	
Individual	Single organism
Cohort	Age class
Population	Collection of individuals making up a gene pool that has a continuity in time due to the reproductive activities within the population (MacLean and Evans 1981)
ESU (Evolutionary Significant Unit)	Population that 1) is substantially reproductively isolated from conspecific populations and 2) represents an important component of the evolutionary legacy of the species (Myers et al. 1998). It is often represented as a spatial area as well.
Species	Interbreeding populations

CHAPTER 8. MANAGING UNCERTAINTY IN SALMON HABITAT RECOVERY PLANNING

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[Note: Most references not yet included.]

To know one's ignorance is the best part of knowledge.

Lao Tzu, The Tao, no.71

Salmon recovery planning requires a complex series of decisions about habitat actions. These decisions and actions are unavoidable despite a large amount of uncertainty in the information available. Inevitably, they will be based on a tapestry of models, estimates, expert opinion, myth, political posturing, predictions, and data. By identifying, quantifying, and incorporating the sources of uncertainty in information used for recovery planning, we can improve the quality of habitat planning. This chapter provides guidance, via examples, for managing the inherent uncertainties of habitat management in a recovery planning context.

A quick example illustrates the importance of identifying and quantifying uncertainty. In choosing between two possible culverts for restoring fish passage, one might be presented with information that culvert A will increase fish capacity by 120 fish while culvert B will increase fish capacity by 100 fish. Without additional information and with no estimates of uncertainty, a manager would surely choose culvert A because it has the highest expected increase in fish capacity. If culvert A opened up habitat that was less certain to be occupied (95% confidence interval = 120 +/- 70), while culvert B opened up wetland habitat that would be quickly colonized with a high degree of certainty (95% confidence interval = 100 +/- 10), a different decision might be reached. Partitioning of the uncertainty might reveal that the primary source of uncertainty about culvert A was the lack of data on juvenile use of a few habitat types because they tended to be inaccessible. Additional information beyond the point estimates of 100 and 120 not only improves decisionmaking but also identifies critical knowledge gaps that can be filled with additional field analysis. We emphasize that all information used in habitat analyses should include both the value and an estimate of the associated uncertainty.

This chapter provides four short examples of uncertainty in habitat management issues related to recovery planning. We describe how management might be improved by acknowledging and incorporating uncertainty in the decisionmaking process. These examples are purposefully simplified. We hope that by deleting site-specific and mathematical details, a general framework for incorporating uncertainty into these decisions can be expressed more clearly. References are provided for each example so that interested readers can locate more detailed information if required.

Example 1: Evaluating a Prediction

Managing uncertainty involves acknowledging, quantifying, and communicating the sources of uncertainty in available information, including predictions on which management decisions will be based. By prediction, we mean a predicted value or a predicted relationship, for example, effects of high flows on egg survival, habitat capacity estimates under different conditions, effects of various restoration techniques, estimates of extinction risk or population trajectories, etc. Decisions can then be based on both the predictions and the uncertainty surrounding them. In this example, we break the uncertainty of a habitat capacity prediction into five classes: predictive uncertainty, parameter uncertainty, model uncertainty, measurement uncertainty, and natural stochastic variation (Table 8-1). We work backwards from a prediction of habitat capacity for a particular watershed to identify the component sources of uncertainty. We provide examples of how each type of uncertainty arises, how it might be quantified, and how it might be reduced. We conclude each subsection with a summary of how decisionmaking can be improved by quantifying and acknowledging each class of uncertainty.

Identification of each potential source of uncertainty is valuable because it tells us where to be skeptical and what additional information or analyses might provide the best reduction in uncertainty or improvement in precision. A series of questions to ask of any prediction is provided in Table 8-2. In some cases, it may turn out that the predictive uncertainty is so large that the available information provides little guidance.

Prediction Uncertainty

Prediction uncertainty includes uncertainty that results from the natural stochastic variation of the system being modeled, measurement uncertainty of the data used to build the model, uncertainty surrounding the form of the model, and parameter uncertainty. In addition, predictions include uncertainty that results from applying the model to a new situation. A capacity prediction for Watershed X might be based on past data for the same watershed or on past data collected in other watersheds. In either case, the prediction is likely to be used for estimating potential future conditions, clearly a new situation. The uncertainty associated with these extrapolations or similar extrapolations from, say, the laboratory to the field, is difficult or impossible to quantify but should be considered and described.

Prediction uncertainty can be assessed to some degree by “ground-truthing” and by simulation studies. In simulations, the model is constructed and parameterized using a subset of the data. The model is then assessed by how well it predicts the subset of data excluded from the model construction. Neither of these methods will remove uncertainty associated with predicting future conditions. A careful assessment of which model components might be sensitive to expected differences between current and future conditions or potential changes in mechanism over time can provide some basis for evaluating the usefulness of modeled predictions. Models can be compared in their relative sensitivity to changing conditions. Models that rely on correlation rather than causation are particularly likely to have high levels of prediction uncertainty.

Parameter Uncertainty

Every parameter in a model is an estimate. Some parameters have a biological interpretation and are therefore like a prediction, for example, increase in habitat capacity per mile of instream wood restoration. Including the uncertainty of these parameter estimates is critical for making good management decisions. If a 20% increase in fish survival for every mile of instream wood restoration is reported, we could sit back quite confidently after restoring five miles. However, if all the information had been presented (20% +/- 30%), we would have been better able to manage the land, perhaps by diversifying the types of restoration actions used, by adding wood to a greater number of stream miles, or by choosing a different restoration action with a smaller but more certain fish response.

For statistical models, these parameter estimates are developed from the data and parameter uncertainty estimates are relatively easy to compute. For mechanistic models, parameters may be estimated from data, from similar models of other phenomena, or by expert opinion. Where parameters are not estimated from data, the uncertainty surrounding them can be very difficult to quantify. Sensitivity analysis can be used to put bounds on parameter estimates and to assess effects of parameter uncertainty. By systematically varying each of the parameters in the model, one can estimate the effect of small changes in parameter values on the model predictions. Where small changes in parameter values lead to large changes in model predictions, it becomes all the more critical to quantify, and if possible to reduce, the uncertainty of parameter estimates. Models that are extremely sensitive to small changes in parameter estimates and which do not have adequate data to estimate those parameters well are not useable management tools. Increased precision of parameter estimates can be achieved by collecting more data or better data (data with less measurement uncertainty).

Model Uncertainty

Nearly all estimates and predictions used in management have an underlying model, either explicitly or implicitly. Uncertainty exists about both the model form (for example a linear relationship versus a Ricker curve) and about which predictor variables to include. In our capacity example, we might have a model that predicts habitat capacity as a linear function of several habitat parameters: wood density, pool density, gradient, adjacent land use, and water temperature. Imagine that this model is fit as a simple linear regression. There is uncertainty as to whether the effects of these five habitat descriptors are additive and have a linear relationship to habitat capacity. The linearity assumption may be valid for the range of, say, water temperature for which we have data, but invalid outside that range. We are also unsure if these five habitat descriptors are the best set of predictors. Why not basin area or flow, for example?

Or why not choose a smaller set of habitat descriptors? Many statistical tools are available for choosing between models (adjusted r-squared, AIC, BIC, f-tests, likelihood ratio tests). In general all these techniques balance the degree to which the model fits the data with the number of parameters required to achieve that fit.

Model uncertainty can have enormous management implications. Models can be wrong because they are not accurately describing the ecological process or because they fail to include an important predictor. If the model is wrong, it is easy for predictions to be of the wrong magnitude and even the wrong direction. In the habitat capacity example, using a linear model for temperature could result in predictions of dramatically increased habitat capacity for temperature reductions from 7°C to 0°C. In ecology, discussion of model uncertainty is rare; yet this is often where managers and ecologists have made big mistakes. Our model of habitat effects on fish survival once assumed that fish survival decreased with increasing amounts of instream wood (ref). Failure to assess the possibility that this model was incorrect (ref) has contributed to habitat degradation in the Pacific Northwest.

Model uncertainty is very difficult to quantify because there are an infinite number of possible models, and in most cases, none are exactly correct. Simulation studies are often used to reduce model uncertainty and examine other things. Simulation studies simulate data from a particular model and then ask questions about the behavior of that data. They can quantify the degree to which model structure impacts predictions. Beyond these tools, reducing model uncertainty is extremely difficult. Schnute and Richards (2001) suggest that model uncertainty be managed by keeping an open mind, identifying all assumptions, and testing assumptions continuously.

Measurement Uncertainty

Measurement uncertainty is simply the difference between the true value and our recorded observation of it. It is directly related to both the accuracy and precision of the measurement technique and data processing. All observations have some level of measurement uncertainty; it can be large and problematic or small and negligible. Some phenomena are inherently difficult to measure and tend to have lots of associated measurement uncertainty, for example, fish survival in different habitats. Other information can be measured quite accurately, for example, stream gradient.

In addition to adding noise or variability, measurement uncertainty can also lead to bias. Bias is a directional error that results from measurement using an inaccurate tool. Biased or potentially biased measurements might include subjective assessments or incomplete records. A less visible form of bias occurs when a measurement technique tends to overestimate in certain conditions and underestimate in other conditions. A simple example is helicopter redd surveys. Redds are easier to identify where there are fewer trees; therefore the accuracy of the measurement depends on whether there are riparian buffers. If the bias is not corrected, the data could end up being used in a model that therefore predicts increases in redd density with removal of riparian trees.

Measurement uncertainty can be reduced but not eliminated. Replication is the best way to reduce measurement uncertainty though it will not remove bias. The best way to manage bias is to estimate it and correct for it. At least some unbiased measurements will be necessary to estimate the bias. In some cases, measurement uncertainty can be very difficult to assess. Expert opinion or subjective assessments, for example, are often used because no actual data exists. Although it may be possible to determine how well experts agree with one another (precision), it is impossible to assess or quantify the relevant issue, which is accuracy.

In some cases measurement uncertainty can be cheap and easy to reduce. By quantifying it, assessments can be made as to the value of collecting more data with the same technique versus using a more expensive technique. Where bias is impossible to measure or quantify, sensitivity analyses (as described in the subsection on parameter estimation uncertainty) can provide an assessment of the degree to which small amounts of measurement uncertainty or bias in the input data might effect predictions.

Natural Stochastic Variation

Natural stochastic variation is the inherent random variability in ecological systems, such as temperature or population fluctuations. It contributes to our inability to make precise predictions. Increased amounts of process uncertainty require increased numbers of observations (either more sites or more replications or both) to make estimates of a given precision (Shea and Mangel 2001). Very high levels of process uncertainty may mean that estimates that are as precise as needed are simply impossible to measure (Korman and Higgins 1997). Identifying and quantifying natural stochastic variation will help us to distinguish between situations in which small amounts of additional data should dramatically increase our ability to make good decisions, and situations in which additional data is unlikely to provide significant increases in accuracy of predictions.

Because stochastic variation is a natural phenomenon, it cannot generally be reduced to increase the precision of our predictions. The best method for reducing its impact on prediction uncertainty is to collect covariates. For example, water temperature may be highly stochastic but we can reduce the noise by stratifying observations with covariates such as time of day and stream size. Beyond the use of covariates, process uncertainty is best managed by simply quantifying and acknowledging it.

In conclusion, evaluating the uncertainty of a prediction requires the dissection of that uncertainty into its components—quantifying and evaluating each. By asking the questions in Table 8-2, we reduce the chances of making poor decisions because of poor predictions. Uncertainty analysis is key to identifying critical knowledge gaps, identifying options for improving prediction precision, and making the best possible habitat decisions.

Example 2: Creating a Prioritized List of Restoration Projects

Prioritizing among potential habitat actions is difficult because of uncertainty about how fish may respond to changes in the environment. For example, we may have a list of potential actions, each of which is expected to increase pool habitat. However, there may be uncertainty about the density of fish that can be supported by a given amount of pool habitat. By explicitly including the uncertainty in a decision table, we can identify the actions with the highest expected final fish density (ref).

The first task is to describe the “alternative states of nature” and ascribe probabilities to these alternative states. In this example, the alternative states of nature are the alternative hypotheses about how many juveniles are supported by a given area of pool habitat. Table 8-3 presents some sample hypotheses and associated probabilities. The probabilities associated with each hypothesis may be generated in a number of ways. One method that can combine multiple types of information is Bayesian meta-analysis, which can pull together information from different studies (ref.). If only limited or ambiguous data are available, expert opinion can be solicited to assign probabilities to the various hypotheses. Numerous texts describe methodologies for soliciting expert opinion (ref). As noted elsewhere in the chapter, it is impossible to know if expert opinion is correct precisely because we use it in situations for which we have no data. If expert opinion is used to assign probabilities to the hypotheses, then the prioritized list that emerges from the decision-analysis process will be a formalization of those opinions.

The next step is to associate an "outcome" with each potential action, assuming each of the alternative hypotheses about the state of nature is true. For example, if the hypothesis that pools can support 5 fish/m^2 is true, then the number of fish expected from the removal of culvert A might be 2,744 fish. In this example the outcome is number of fish, but other appropriate outcome units could be selected (such as fish/$). This outcome is calculated based on an assessment of the number of pools that would be made available after removal of the culvert. More realistic and detailed decision tables might also include additional information such as the number of riffles, the types of pools, the depth of the pools, or the quality of the expected pool habitat. Table 8-4 shows potential outcomes, in total fish, for a number of potential management actions as a function of fish density in pools.

Finally, we can calculate the final expected outcome of each of the potential actions, given the probabilities of the states of nature (Table 8-3). The final expected outcome of each action can be calculated as the sum of the expected outcomes for each state of nature (Table 8-4) times the probability that the state of nature is true (Table 8-3). For example, the expected outcome for removal of culvert A is 2744 * 0.1 + 4892 * 0.3 + 5248 * 0.5 + 5786 * 0.1 = 4945. Table 8-5 shows the expected outcome for each of the four potential actions. Removal of culvert A would be expected to provide the largest increase in total number of fish.

This is an extremely simple example. Hypotheses about the states of nature will often involve more than a single dimension (i.e., more than only pool density). Many types of information can be included in the analysis, but there will often only be one or two critical uncertainties that drive a decision. Decision tables provide a structured method for including and communicating uncertainties. Clearly communicating the decision path is one of the greatest values in constructing decision tables. Decision tables can easily be constructed for many of the examples in this document. For example, the methods for prioritizing restoration actions described in Chapter 5 could be modified to include uncertainty about fish response, restoration costs, or habitat quality by using the decision table methodology described here.

Example 3: Water Quality and Salmon Health, Uncertainty Across Multiple Scales of Biological Complexity [incomplete]

We have, at best, an incomplete understanding of chemical habitat quality in all but the most pristine river systems in the Pacific Northwest. The absence of environmental monitoring data for many contaminants is a critical source of uncertainty. If baseline environmental conditions have not been defined, it can be very difficult to estimate the impacts of pollution on the health of salmon or the integrity of aquatic food webs. The most difficult challenge facing natural resource managers is perhaps the sheer number of different chemicals that human activities deliver to river systems and estuaries. The list of chemicals that are potentially toxic to salmon or aquatic ecosystems is very long. It includes chemicals such as pesticides, metals, surfactants, PCBs, dioxins, hydrocarbons, plasticizers, detergents, pharmaceuticals, and various industrial solvents. These contaminants occur in complex mixtures that vary considerably in time and space. Salmon habitat conditions may reflect current land use activities or, for the more persistent chemicals such as DDT, activities that were restricted or banned many years ago.

Recovery plans that are structured around physical processes may fail to capture important spatial patterns of chemical habitat degradation. For example, water quality at a specific point within a watershed may be determined by land use activities that are far removed from the focus of a restoration effort. For persistent or bioaccumulative pollutants such as PCBs and DDTs, salmon can integrate chemical habitat quality throughout their entire geographic range. A series of exposures to persistent chemicals will ultimately determine a salmon's accumulated toxic burden and, in turn, the likelihood that a critical threshold for the health of the animal will be exceeded. In a sense, migratory salmon carry a legacy of site-specific pollution as they move on to new environments. This creates biological linkages between chemical habitats at scales that may exceed the scope of a recovery plan.

The temporal determinants of chemical habitat quality are also complex. On the one hand, surface water runoff can impact stream conditions on a time scale of hours to days. Conversely, persistent chemicals can degrade sediments for decades or longer. Chemical inputs to rivers and coastal ecosystems are dynamic and can change in a matter of weeks or months. For example, hundreds of different pesticides are now in use in the Pacific Northwest. The site-specific applications of pesticides will vary as cropping patterns change, new chemicals are phased in, and old chemicals are phased out. Consequently, the specific functional relationships between water quality and salmon health can be expected to change from season to season and year to year.

The Importance of Biological Scale

Pollution is an important habitat concern for the vast majority of threatened and endangered salmonids in the Pacific Northwest. Accordingly, habitat-based models for salmon recovery should capture the biological significance of water and sediment quality. There are several reasons why they usually don't. First, "chemical habitat quality" can be very difficult and expensive to define. Second, there is a general absence of toxicological data for most of the chemicals that have been detected in salmon habitat. Only a few studies have specifically addressed the impacts of these contaminants on salmon health or on the integrity of aquatic communities in the Pacific Northwest. Third, effects at the scale of natural populations are rarely considered in a conventional toxicological study. Many endpoints, or biomarkers of exposure, have no clear or consistent relationship to the survival or reproductive success of the exposed animal. Consequently, there is often a disconnect between the biological scale of toxicological studies and current habitat recovery models.

The legacy factor — the adverse impacts of degraded water or sediment quality may not immediately manifest at a particular site. Instead, key systems may fail at a later and critical life history stage (New Brunswick nonylphenol example).

The integration factor — a particular concern for bioaccumulative chemicals, since salmonids are highly migratory. Salmon are, in certain respects, integrators for habitat quality throughout their geographical range, and the adverse affects (or threshold) of a persistent chemical depends on the total accumulated body burden. Therefore, there can be important linkages in chemical habitat quality at spatial scales that are beyond the scope of most habitat recovery plans. Mercury, DDTs, PCBs....

Reconciliation

Despite all of the uncertainty, there are ways to establish cause and effect relationships that are meaningful at biological scales used in habitat recovery planning. Explain. Use these in the decision process.

Example 4: Inadequate Information for Making a Decision

A careful and honest examination of uncertainty in data, predictions, and models will inevitably lead to the identification of many situations in which adequate data for making a data-based decision are simply not available. We strongly discourage basing the decision on biased or imprecise predictions, prioritization systems for which guesswork must be substituted for data, or information that becomes inaccurate or imprecise at the scale for which the decision must be made. Instead, we suggest explicitly providing a rational for the decision that requires minimal data.

The most important characteristics of a decision rule are that it is can be documented and that it is robust. Documentation is important because future managers will need to understand the basis for the decision. This requirement prevents arbitrary decisions in the face of inadequate data. Decision rules that are robust to the uncertainties in the information prevent risky management decisions (Schnute and Richards 2001). Decision rules that have been presented in the literature include the following two examples.

The Precautionary Principle can be defined as "when an activity raises threats of harm to public health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically" (ref). Because this principle shifts the burden of proof to those who create risks and does not define which risks are most important (Hilborn et al. 2001), it has generated much controversy. However, there are many examples of national and international policies that have been based on the precautionary principle. European environmental law is based on the precautionary principle through the Treaty on European Union, 1992 (Foster et al. 2000), and the United States is bound by international treaties, including the Rio Declaration from the United Nations Conference on Environment and Development, to implement the precautionary principle in environmental health protection (Raffensperger and Tickner[year]). While we are not advocating this particular decisionmaking rule, we present it as an example of a relatively simple guiding principle by which high-level decisions can be made before adequate data are available.

Safe minimum standard (SMS) is another decisionmaking rule that has received considerable attention. The SMS approach is a collective choice process that prescribes protecting a minimum level of a renewable resource unless the social costs are excessive (Berrens 2001). This approach to making environmental decisions is usually invoked in settings involving considerable uncertainty and potentially irreversible losses. Again, there are multiple potential sources of controversy. We present this approach for comparison to emphasize the importance of carefully choosing the decisionmaking principle and documenting exactly what considerations should be involved. The choice of a guiding principle will dictate management decisions until improved information is available.

The choice of a decisionmaking rule need not be so theoretical. Chapter 2 discussed the need to define a habitat strategy that includes gathering additional data and taking interim actions. These interim actions are an excellent example of a using a guiding principle until adequate data become available. Chapter 5 presented guidelines for selecting restoration actions before all of the habitat data is available. This is a simple and effective method for dealing with incomplete information.

In conclusion, we emphasize that inadequate data does not need to prevent decisions from being made. It is possible to implement strategies that require minimal data. Such strategies are preferable to using biased or imprecise predictions, guesswork disguised as data, or information that is inappropriate to the scale of the decision. Applying such a strategy carries with it two obligations. First, we must try to improve the information available for the decision in the future. The analyses described in the first example of this chapter (evaluating a prediction) can be used to identify and reduce critical information uncertainties. Second, we must set a time frame for reevaluating the decision. In the best possible scenario, decision strategies that require minimal data are used simply as interim measures.

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Table 8-1. Tools and methods for quantifying and reducing uncertainty.

Class of uncertainty	Brief definition	Habitat example	Methods for quantifying	Possibilities for reducing
Prediction uncertainty	Difference between the modeled response and the true response.	Uncertainty of predicting habitat capacity of a given watershed after instream restoration.	Leave-one-out estimates of prediction error rates. Simulation studies comparing conditions where model was built to those in which it is being applied.	Collect data for conditions in which predictions are required. Do not extrapolate beyond conditions under which model was developed.
Parameter uncertainty	Difference between the true parameter (such as an average or a regression coefficient) and the parameter as estimated from the data.	Uncertainty of parameters describing change in capacity as a function of changes in watershed condition.	Statistical theory for model coefficients derived from data. Sensitivity analysis for model coefficients estimated from other sources.	Collect more data or more accurate data. Collect data over a wider variety of conditions.
Model uncertainty	Difference between natural system and the mathematical equation used to describe it. Includes model form and set of predictors.	Uncertainty in the relationship between habitat conditions and fish capacity. Uncertainty in which habitat descriptors are the best predictors of fish capacity.	Statistical descriptions of model fit: AIC, BIC, likelihood ratios, F-statistics.	Consider wide variety of models.
Measurement uncertainty	Difference between true value and the recorded value.	Uncertainty in measurements of data used to build the predictive model, i.e. fish or redd density under differing habitat conditions.	Test accuracy of measurement technique against standard method or known values.	Improve measurement techniques. Increase the number of replicates. Calibrate biased measurement techniques.
Natural stochastic variation	Inherent random variability.	Natural fluctuations in population size, habitat selection, or habitat	Variance of the observed data. Variance of the observed	Collect covariates. Collect more replicates for conditions of interest.

	conditions.	data for different sets of conditions.	

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Table 8-2. Questions to guide the evaluation of predictions.

Prediction uncertainty
How similar are the conditions under which the original information was gathered to those for which the prediction is being made?
How sensitive is the model (data, mechanism, and parameter estimates) to site-specific details?

Parameter uncertainty
Is the prediction sensitive to small changes in parameter estimates? If so, how precise are the estimates of those parameters?

Model uncertainty
What are the assumptions on which the prediction is based? How sensitive is the prediction to these assumptions?

Measurement uncertainty
Could any of the information on which the prediction is based be biased?

Table 8-3. Example alternative hypotheses about the states of nature (i.e. density of fish per m^2 of pool habitat). The probabilities describe the relative likelihood that each hypothesis is true. All probabilities must sum to one.

Hypothesized fish density	5	10	15	20
Probability hypothesis is correct	0.1	0.3	0.5	0.1

Table 8-4 Expected outcomes for potential habitat actions (total fish) as a function of hypothesized fish density per pool.

Potential action	Hypothesized fish density per pool			
	5	10	15	20
Remove culvert A	2744	4892	5248	5786
Remove culvert B	2844	3400	3858	6457
Remove riprap	2012	4172	4260	4340
Add wood	1568	3410	5963	6230

Table 8-5. Overall expected outcomes (increase in total number of fish) of potential actions.

Potential action	Expected outcomes
Remove culvert A	4945
Remove culvert B	3879
Remove riprap	4017
Add wood	4784

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GLOSSARY

Anadromous

Moving from the sea to fresh water for reproduction.

Anthropogenic

Caused or produced by human action.

B-IBI

Benthic Index of Biological Integrity

Biodiversity

Biological diversity in an environment as indicated by numbers of different species of plants and animals.

Biological integrity

Biota

The flora and fauna of a region.

Channel width (wetted width and bankfull width)

Bankfull width is the channel width between the tops of the most pronounced banks on either side of a stream.

Culvert

Buried pipe or covered structure that allows a watercourse to pass under a road or underground.

CWA

Clean Water Act

Distributary (also, distributary channel)

A branch of a river or stream that flows away from the main channel and does not rejoin it.

Disturbance

Introduction of an unwanted condition into a system or interference with a habitat's normal or existing conditions.

Ecoregion

An area determined by similar land surface form, potential natural vegetation, land use, and soil; it may contain few or many geological districts (Omernik 1986).

Ecosystem

The system of all the living and dead organisms in an area and the physical and climatic features that are interrelated in the transfer of energy and material.

EIA

Effective impervious area

Endangered species

A species in danger of extinction throughout all or a significant portion of its range.

ESA

Endangered Species Act

ESU

Evolutionarily significant unit**:** An ESU is "a population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species (Myers et al. 1998)." It is sometimes represented as a spatial area as well.

Field data (compare to remotely sensed data)

Floodplain

Lowland area bordering a stream, onto which it spreads at flood stage.

GIS

Geographical information system

Glide

Relatively slow and shallow stream section with water velocities of 10-20 cm/s and little or no surface turbulence.

Hydromodification

IBI

Index of biological integrity

Impoundment

Interim strategies and actions

Landsat

Landsat satellites supply global land surface images.

LWD

Large woody debris: Any large piece of woody material that intrudes into a stream channel.

Main stem

Principle stream or channel of a watershed system.

Multi-species management

Outlier

In statistics, any data point exhibiting anomalous behavior.

Population

A group of individuals of a species living in a certain area that maintain some degree of reproductive isolation.

Pool:riffle:glide ratio

Ratio of the respective surface areas or lengths of pools to riffles to glides in a given stream reach, often expressed as the relative percentage of each category.

Reach

Recovery

Redd

Nest in gravel, dug by a fish for egg deposition, and associated gravel mounds.

Refugia

Remotely sensed data (compare to field data)

Data gathered at a remote station or platform, as in satellite or aerial photography.

Restoration

Riffle

Shallow section of a river or stream with rapid current and a surface broken by boulders, rubble, or gravel.

Riparian

Relating to or situated on the area between a stream or other body of water and the adjacent upland.

Riprap

Layer of large, durable materials used to protect a stream bank from erosion; may also refer to the materials used, such as rocks or broken concrete.

Salmonid

Fish of the family Salmonidae, including salmon, trout, and chars.

Seral

Of, relating to, or constituting an ecological sere. (A sere is a series of ecological communities formed in ecological succession.)

Side channel

Smolt

Juvenile salmonid in its seaward migrant stage.

SMS

Safe minimum standard

Species

Stochastic

Of or relating to uncertainties or random variables.

SWAM

Salmonid Watershed Analysis Model: a large scale landscape analysis for identifying high priority areas for salmon habitat restoration.

Taxa

Plural of taxon, a taxonomic group or entity.

TRT

Technical Recovery Team: The TRT is responsible for establishing biologically-based ESA recovery goals for listed salmonids within a given recovery domain. Members serve as science advisors to the recovery planning phase.

Threatened species

A species not presently in danger of extinction but likely to become so in the forseeable future.

VSP

Viable Salmonid Population

WDFW

Washington Department of Fish and Wildlife

Watershed (also basin or catchment area)

The drainage area of a stream.

*****FOR REVIEW ONLY*****

APPENDICES

[**For** *Ecosystem Recovery Planning for Listed Salmon: An integrated assessment approach for salmon habitat*]

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*****FOR REVIEW ONLY*****

APPENDIX A. RESTORATION OF HABITAT-FORMING PROCESSES: AN APPLIED RESTORATION STRATEGY FOR THE SKAGIT RIVER

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Abstract

The Skagit Watershed Council adopted a salmon habitat restoration strategy that strives to identify: 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and habitat conditions. Therefore, habitat restoration and protection actions are directed at restoring the disturbed habitat-forming processes instead of attempting to build specific habitat conditions. In the first phase of applying the strategy, the Council uses existing GIS data and field-based inventories to identify disturbances to five different landscape processes. The second phase relies solely on field-based inventories to identify disrupted processes. We find that 23% and 46% of the 8,544 km^2 Skagit River basin is likely impaired with respect to peak flow and sediment supply, respectively. Field-based inventories have identified 328 km of channel blocked from fish use and 130 km of stream corridor in need of riparian restoration with about a third of the basin inventoried. These analyses have led to identify more than 400 individual restoration projects and been completed in one year. Total cost of developing the restoration plan (both first and second phases) is projected to be less than 1% of the projected restoration costs.

Key Terms: Salmon Habitat Restoration, Aquatic Ecosystems, Watershed Management

Introduction

Escapement levels of Pacific Northwest and British Columbia salmon stocks have declined dramatically in the past century due to habitat loss, high levels of harvest, and changes in ocean conditions. Land-use induced freshwater habitat losses were associated with the decline of nearly all of the stocks at-risk in a recent study by Nehlsen et al. (1992). However, the recognition of the causes of these declines and the desire to restore salmon runs has not led to specific plans for recouping habitat losses in large watersheds. Rather, most habitat restoration actions have been conducted in an unplanned and uncoordinated fashion.

In 1997, the Skagit Watershed Council (SWC) was formed to support the voluntary restoration and protection of salmon habitats in the Skagit River basin of Washington State. Today the Council is comprised of 36 member organizations including private industrial and agricultural interests, state and federal agencies, local governments, tribes, and environmental groups. In 1998, the Council adopted a salmon habitat protection and restoration strategy that recognizes the influence of land use and resource management activities on natural landscape processes, which result in changed habitat conditions (SWC 1998). Since 1998, members of the SWC have completed an interim application of the strategy, which identifies causes of degraded habitats in the watershed and restoration actions that are needed to restore habitats over the long term. In this paper we briefly describe the strategy and methods, and present the preliminary findings of the analyses. We also discuss costs of the analysis relative to projected costs of restoration actions.

Overview of the Restoration Strategy

Figure A-1 is a conceptual diagram illustrating how watershed controls (ultimate and proximate) and natural landscape processes combine to form various habitat conditions. Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas (>1 km^2), and shape the range of possible habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (≤ decades), act over smaller areas than independent controls, and are partly a function of independent factors (Naiman et al. 1992). Landscape processes are typically measured as rates and characterize what ecosystems or components of ecosystems do. For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time period) at which sediment or water is supplied to and transported through specific locations of a watershed. Some riparian related functions can be viewed similarly. For example, large woody debris (LWD) “recruitment” is synonymous with the idea of supply while LWD “depletion” is the result of both LWD transport and decay rates. Natural rates of landscape processes are here defined as those that existed prior to widespread timber harvest, agriculture, or urban development.

The Skagit Watershed Council’s habitat protection and restoration strategy describes a scientific framework and set of procedures for identifying and prioritizing activities that restore or protect aquatic habitat (SWC 1998). The scientific framework strives to identify: 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and habitat conditions. It focuses not on the symptoms of watershed degradation but on the fundamental causes, and encourages restoration and protection of natural landscape processes that formed and sustained the habitats to which salmon stocks are adapted. Justification of this approach is based on our understanding from current literature that natural landscape processes create and maintain the "natural" habitat conditions in which native aquatic and riparian species have adapted (e.g., Peterson et al. 1992, Doppelt et al. 1993, Reeves et al. 1995, Ward and Stanford 1995, Beechie et al. 1996, Kauffman et al. 1997).

We apply the strategy by systematically identifying land-use disruptions to landscape processes that form salmon habitat. These processes include peak flow hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality. Using a series of diagnostic screens, we locate disturbances to habitat-forming processes, and identify actions (i.e., projects) required to correct the disturbances. This paper reports the progress made by the Skagit Watershed Council in applying its strategy within the range of anadromous salmonids of the Skagit River basin in northwest Washington.

Study Area

The Skagit River basin drains approximately 8,544 km^2 of the North Cascade Mountains of Washington State and British Columbia (Figure A-2, section A). Elevations in the basin range from sea level to about 3,275 m (10,775 ft) on Mt. Baker. Numerous peaks in the basin exceed 2,500 m in elevation. Average annual rainfall ranges from about 90 cm (35 in) at Mt. Vernon on the lower flood plain, to over 460 cm (180 in) at higher elevations in the vicinity of Glacier Peak. Several vegetation zones occur in the study area. As defined in Franklin and Dyrness (1973), most of the lower elevations are in the western hemlock zone. These forest zones typically include western hemlock (<u>Tsuga heterophylla</u>), Douglas fir (<u>Pseudotsuga menziesii</u>), western red cedar (Thuja plicata), and sitka spruce (Picea sitchensis). Deciduous species in this zone include red alder (Alnus rubra), black cottonwood (Populus trichocarpa), and big leaf maple (Acer macrophyllum). Middle elevations are in the Pacific silver fir (Abies amabilis) zone, and higher elevations are in the alpine fir (A. lasiocarpa) zone (Franklin and Dyrness 1973).

The Skagit River basin is comprised primarily of mountain drainages, with fewer lowland sub-basins (low topographic relief and low elevation). The hydrographs of most low-elevation forested sub-basins are dominated by autumn and winter rainfall floods (Beechie 1992). Conversely, spring snowmelt floods typically dominate the hydrographs of high elevation sub-basins in the eastern Skagit. Most areas of the Skagit basin are of intermediate elevation and exhibit both rainfall and snowmelt floods. Lowland sub-basins are generally located in the western valley (rain dominated) and are usually more highly developed by urban and agricultural land use than the forested mountain basins (Lunetta et al. 1997).

Land development (primarily logging and draining or clearing lands for agriculture) began around 1860. About 1590 km^2 (615 mi^2, 19%) of the basin is currently in private and State of Washington ownership. Land uses are dominantly agricultural and urban in the lower flood plain and delta areas, and upland areas are generally commercial forests. About 3680 km^2 (1420 mi^2, 44%) of the basin lies within the federally-owned North Cascades National Park, Mt. Baker and Ross Lake National Recreation Areas, and Glacier Peak Wilderness Area; the U.S.D.A. Forest Service controls an additional 1960 km^2 (755 mi^2, 24%) of the basin in the Mt Baker-Snoqualmie National Forest. Approximately 1040 km^2 (400 mi^2, 13%) of the basin is in the Province of British Columbia.

Anadromous salmonids indigenous to the basin include chinook salmon (Oncorhynchus tshawytscha), coho salmon (O. kisutch), pink salmon (O. gorbuscha), chum salmon (O. keta), sockeye salmon (O. nerka), steelhead trout (O. mykiss), cutthroat trout (O. clarkii), and native char (Salvelinus sp). Access to anadromous fishes is generally confined to elevations below 700 m by natural barriers. Upstream migration to the Baker River system has been eliminated by the installation of two hydroelectric dams, but anadromous fish production - primarily coho and sockeye salmon - is maintained through trapping and hauling operations, in addition to the maintenance of sockeye spawning beaches and smolt bypass trapping. The extent of salmon upstream migration in the Skagit River basin is shown in Figure A-2, section B.

Methods

We analyze disturbances to watershed processes in the Skagit River basin in two phases. In the first (interim) phase, we locate disturbed habitat-forming processes using a combination of existing Geographic Information System (GIS) data and field-based inventories to identify disturbances to peak flows, sediment supply, riparian functions, channel-floodplain interactions, blockages to salmon migration, and water quality. The second phase (to be largely completed during the next two years) relies solely on field-based inventories. Both phases rely on GIS to analyze and maintain landscape process data over the 8,544 km^2 area of the Skagit River basin. This paper describes only the methods and results from Phase 1.

We have used more than 30 different GIS themes and partial field inventories to apply the landscape process screens identified in the Strategy. The existing GIS themes provide low-resolution data covering the entire river basin. These data give us a good overview of habitat-forming processes in the entire basin, but can give erroneous answers to our questions about specific reach level sites (10^2-10^4 meters linear scale). Field inventories provide high-resolution data, but with only limited coverage at present. Because field-inventories are more reliable at specific sites, the Skagit Watershed Council members have made a long-term commitment to collecting field-based information basin-wide.

We analyzed selected landscape processes that form salmonid habitats in the Skagit River basin. We selected these analyses based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes (Table A-1). We recognize that the list may not include all impacts to salmon in the watershed. However it includes those that are clearly supported by scientific literature and are responsible for a significant proportion of the total loss in salmon production from the basin. For each process we developed a series of diagnostics based on rates derived from scientific literature and local studies. The diagnostics and methods for estimating values are summarized in Table A-1.

We synthesized the ratings for individual landscape processes and functions into a single reach classification that we call the generalized habitat types. The importance of identifying generalized habitat types for watershed restoration is illustrated by Frissell (1993) and Doppelt et al. (1993), where examples of habitat types are listed along with their biotic objective and restoration tactics. To apply this concept in the Skagit, we derived generalized habitat types based on simple correlations between our understanding of anadromous fish life history strategies and reach level habitat types (approximately 10^2 to 10^4 meters in linear scale) (Table A-2). We assume that relationships between fish life stages and habitat for each indicator species analyzed adequately identifies the "habitats to which salmon stocks are adapted" in an effort to be consistent with our goal.

Our analysis used five species and four life stages to determine generalized habitat types. The life history stages examined were 1) spawning/egg to fry, 2) summer rearing, 3) winter rearing, and 4) estuary rearing. Several salmonid species present in the Skagit River basin were excluded from the evaluation due to lack of data or a spatial bias in their distribution not related to geomorphic habitat types. Native char were excluded due to a lack of data describing their habitat preferences over their complete life history. We know the spawning range of native char is bias toward higher elevation headwater tributary basins which is included in the range of historical anadromous fish access (Figure A-1, section B). Cutthroat trout were excluded because of their spatial bias towards the lower elevation rain-dominated sub-basins of the Skagit. Coho salmon habitat preference is similar cutthroat, and the coho range includes all of the anadromous cutthroat range in the Skagit, therefore we assume that coho relationships in our analysis adequately represent cutthroat. Sockeye were excluded because the population is limited to one sub-basin the Skagit: the Baker River. While resident rainbow trout (O. mykiss) are found throughout the entire river basin, they are assumed to have the same juvenile habitat preferences as steelhead (the anadromous form of O. mykiss) which are included in our analysis.

Under pristine habitat conditions (i.e., natural disturbances only) we define reach-level habitat types for anadromous salmonids in the Skagit as either “key” or “secondary” (Table A-2). Key habitat is “critical” for at least one life stage or is a “preferred” habitat type by the majority of life stages considered. Secondary habitat does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered. Classification systems described in Hayman et al. (1996), Montgomery and Buffington et al. (1997), Peterson and Reid (1984), and Simenstad (1983) were used to define the different reach level habitat types. Local studies used to designate whether the specific reach level habitat types were “critical”, “key”, or “secondary” for a life history stage included: Beamer and Henderson (1998), Beechie et al. (1994), Congleton (1978), Congleton et al. (1981), Hayman et al. (1996), Montgomery et al.

(1999), and Phillips et al. (1980, 1981). Data from outside the Skagit (Queets River, Washington in Sedell et al., 1984) were also used to help understand juvenile fish use differences between large main channels and off-channel habitats.

Under disturbed habitat conditions (i.e., both human and natural disturbances) we designated reach-level habitat types as: "key" when all landscape screening results were rated as functioning; "important" when at least one landscape screen is moderately impaired; "degraded" when at least one landscape screen is impaired; "secondary" when channel type is step-pool or steeper; "isolated" when upstream of a manmade barrier to fish migration, or "unknown." Some reaches are designated as "unknown" because of the high probability of error in rating the riparian condition correctly by land cover types.

Mainstem areas with any of the following conditions were consider degraded: riparian buffer is less than 20 meters wide, streambank edge is hardened (e.g., riprap), or levee is present within 60 meters of the bankfull channel edge. All other Lower Skagit mainstem areas were considered important. In the estuary, areas that are hydromodified are considered degraded, areas adjacent to levees are considered important, areas at least one distributary channel away from a levee are considered key.

Results and Discussion

Hydrology – Changes in Peak Flow

We estimate 23% of the mountain sub-basins in the Skagit are very likely impaired or likely impaired with respect to peak flow hydrology (Figure A-3, section A). In lowland basins, we estimate 7% of the streams historically accessible to anadromous salmon will be impaired when urban and residential areas are fully built out, and 18% will be moderately impaired (Figure A-3, section B). We use the results shown in Figure A-3 to help evaluate the likelihood of success of proposed restoration projects. In general, we do not support restoration efforts directly in or adjacent to channels that are classified as “impaired” without evidence demonstrating that the proposed work will not fail biologically or physically due to the likely increase to peak flows. We also use the results to identify areas currently in good condition that are planned for future development to an extent that hydrology is likely impaired. For these areas we consider protection actions such as rezoning to a less intensive land use or acquisition. We also identify areas to investigate for potential restoration of hydrologic processes.

Reduced peak flows as a result of flood control change a channel’s ability to create and maintain the suite of diverse flood plain habitats to which aquatic species are adapted (Ward and Stanford 1995). Annual peak flows in the Skagit River basin have changed since flow regulation through the construction of reservoirs capable of flood storage. Before flood storage capability, floods in the lower Skagit River commonly approached or exceeded 5,500 cms, and floods in water years 1815 and 1856 were estimated at 11,327 and 8,495 cms, respectively. Since the advent of flood storage capability, a flood approaching 5,500 cms has not yet occurred. The number of floods between the 2-year and 100-year return period has been reduced by roughly 50% since dams were built on the Skagit and Baker Rivers (Table A-3). Flood storage on the Skagit has likely impacted channel-flood plain processes in reaches downstream of the dams, but we have not yet quantified the effects. Until we have a better understanding of these impacts, we view the dams as watershed controls (i.e., as shown in Figure A-1). That is, they operate independently of our management control because they are licensed for up to 50 years and are unlikely to be removed. Accepting that this disturbance will not likely be altered during the license period of each dam, the artificial creation of off-channel habitat then may be justified in stream reaches where off-channel habitat has been lost due to this disturbance. Alternatively, it may be possible to re-establish certain channel-forming flows that have been eliminated in the past.

Sediment Supply

We estimated that 46% of the area in mountain sub-basins of the Skagit is impaired with respect to sediment supply (Figure A-4, section A). Our evaluation of the accuracy of the method showed that it correctly estimated the sediment supply rating for seven of the ten sub-basins where sediment budget data were available. It over-estimated average sediment supply for two of the ten test basins (i.e., rated them impaired when they are functioning), and under-estimated sediment supply for one sub-basin. Therefore, we recognize that this product should not be used to identify potential restoration projects. Rather, it is used for project screening where field-based sediment budgets are not available, and for general planning watershed-level sediment reduction projects. For project screening, project proponents use this map to determine whether the proposed project area is likely to have an impaired (i.e., high) sediment supply. For reaches where sediment supply is impaired, 1) sediment supply in the watershed should be restored to "functioning" levels before downstream reaches are worked on, or 2) evidence demonstrating that the proposed work will not fail due to increased sediment supply should be presented.

Specific sediment reduction projects are identified based on the results of forest road inventories. We focus on forest roads for sediment reduction projects because mass wasting rates from forest roads averaged about 44 times more than mass wasting rates in mature forest

(Paulson 1997). Currently, about 1,300 km of road are inventoried with another 3,000 km remaining (Figure A-4, section A). Risk ratings from the current inventory showed that a significant number of forest roads in the Skagit basin pose a landslide hazard and potentially threaten fish habitat. Based on this inventory, we will focus initial sediment reduction projects on the high-risk and moderate-risk road segments.

For example, the Bacon Creek watershed has 3.7 km and 18.6 km of high and moderate risk roads respectively (Figure A-4, section B). The high-risk road segments cross relatively more of the landforms sensitive to disturbance for mass wasting (e.g., inner gorges and steeper hillslopes) than moderate or low risk roads. Sediment reduction projects on these roads would reduce the risk of sediment supply increasing due to roads therefore increase the level of watershed protection. Specific road projects primarily involve stabilizing or re-contouring road fills, removing stream crossings or improving drainage conveyance, and improving road surface drainage. For river basin level planning, we consider sub-basins with the lowest total cost of road restoration per kilometer of salmon stream as the highest priority.

Riparian Function

Before interpreting the Landsat classification of riparian forests, we used field inventory results from 234 riparian sites to describe the distribution of field-based riparian classifications within each satellite-based forest class (Table A-4). All of the sampled “late-seral forest” sites, and between 92% and 88% of the “mid-seral forest” and “early-seral forest” sites met the > 40 m wide riparian buffer criteria, fitting our functioning designation. Conversely, 90% of the areas mapped as “non-forest” had < 20 m wide riparian buffers, fitting our impaired designation.

Areas mapped as "other forest" (ranging from clearcuts to mature hardwoods) were found to be 43% functioning, 15% moderately impaired, and 42% impaired.

Based on this analysis, we estimate that 29% of the non-main stem channels in the anadromous zone (by length) are in the non-forest land cover category, and therefore have a very high likelihood of being impaired and in need of riparian restoration (Figure A-5). Conversely, 19% of the non-main stem channels in the anadromous zone are in the mid to late seral forest land cover category, and therefore have a high likelihood of being functioning and therefore needing protection. While we can not accurately map stream reach scale riparian conditions associated with channels adjacent to all GIS land cover types, we can estimate with reasonable accuracy the total of each riparian category at a larger scale. Based on the results in Table A-4, we estimate the percentage of non-main stem channel length in the Skagit anadromous zone by each land use designation (Figure A-6).

We rely on field inventories to identify actual restoration projects because of the above mentioned limitations in the satellite classification of riparian forests. We conducted field inventories of riparian forests by walking all streams accessible to anadromous fish and assessing the riparian vegetation conditions for each stream reach. We classified riparian conditions by buffer width, stand type, and age of vegetation within 60 meters of stream channels. From these data, we selected all stream segments with forested riparian vegetation less than 40 meters wide as requiring planting, and all segments with evidence of livestock access to the stream channel as requiring fencing. Riparian planting and restoration projects have been identified through a series of field inventories. The inventories were completed systematically as four separate projects between 1995 and 1998 in 24% of the Skagit's sub-basins (Figure A-5). Together, the inventories identified 130 km of stream corridor for riparian planting and fencing projects.

Isolated Habitats and Disrupted Channel-Floodplain Interactions

The inventory efforts through September 1999 have identified 229 manmade barriers out of 572 channel crossing structures with 32% of the anadromous zone inventoried. In tributary habitat, 143 km of channel is blocked. In the delta, we estimate 185 km (56%) of the channels have been isolated or lost to salmon access under present conditions (Figure A-7). Isolated channels are those where a topographic channel and water exist, but juvenile or adult salmon access is blocked due to man-made disturbances. Lost channels are those areas that were historically channels, but currently do not have clear a topographic channel or water present.

Because manmade barriers are not evenly distributed throughout the Skagit River basin and our inventory efforts have focused in areas where barriers are more common, we anticipate that the majority of the isolated habitat in the basin has been found. Based on a sub-sample of 111 inventoried structures within sub-basins of the Skagit with similar land-use intensity as the sub-basins yet to be inventoried, we found that 14% of the inventoried structures do not meet fish passage criteria. Therefore, we expect to find about 150 more blockages in non-inventoried areas of the basin, blocking about 60 km (4%) of the estimated length of tributary habitat in the anadromous zone.

Upstream of the Skagit delta, 46 kilometers of stream banks have been rip-rapped (Figure A-7). In the geomorphic delta, 51 km (62%) of the mainstem channel edge is either hardened, diked within 60 meters of the channel's edge, or both. These inventory results provide the basis for identifying potential rip-rap removal (or modification) projects, primarily where hardened banks no longer protect capital improvements (e.g., house, road).

Generalized Habitat Types

The final result of our analysis is the identification of generalized aquatic habitat types throughout the entire river basin, which are based on salmonid habitat preferences combined with the results of the landscape process screens. The resulting analysis in the Skagit basin yields a mosaic of reach-level habitat patches (Figure A-8). Key habitat areas have all habitat-forming processes functioning at or near historic levels, and are targeted for habitat protection. Because protection of habitat is generally considered less expensive than restoration, we view key habitats as some of the highest priority areas for habitat expenditures. Isolated habitats are typically the most cost-effective restoration projects, and therefore receive strong consideration for restoration funding. Important and degraded habitats are both areas that are targeted for restoration.

Secondary habitat will not be targeted for restoration under this strategy. That is, we do not intend to "restore" secondary habitat to key habitat. However, it is important to understand how secondary habitat may function in a watershed in order to protect or restore the other habitat types. For example, the source of degradation may originate in secondary habitat (i.e., the idea of contributing critical areas, discussed in Frissell, 1993). In such cases, restoration of processes originating in secondary habitat areas may be required in order to restore downstream degraded or important habitats.

Identification of Restoration Projects

The main objective of the strategy is to identify habitat protection and restoration projects based on application of the landscape process screens. Together, our analyses have led to the identification of more than 400 individual restoration projects. For example, our analysis of the U. S. Forest Service road inventory identified approximately 650 km of high-hazard and moderate-hazard roads that are candidates for restoration. The total estimated cost for all of these roads (which does not include forest roads on state and private lands) is approximately \$11.6 million. We also identified 122 riparian planting and fencing projects during inventories of only 24% of the river basin, with a total cost estimated at \$1,687,000. Of these riparian projects, 39 are already funded.

We completed migration barrier inventories in 13 out of 38 sub-basins, and identified 229 blocking structures. Some blockage removal projects have uncomplicated designs and relatively clear benefits. These projects can each be considered independently of other culverts because salmon currently access the culvert sites, and repair of the structures will provide benefits commensurate with the amount of habitat upstream. By contrast, groups of completely blocking structures on the same water course should be considered either in combination or sequentially, and projects that involve flood protection levees or coordination of numerous landowners require feasibility studies to determine suitable restoration actions. Currently we have a list of 36 isolated sloughs and blind tidal channels that require further assessment for design of appropriate solutions.

Each restoration project is mapped on a GIS theme, and relevant data are stored in the associated databases. These themes can be updated as new inventories are completed, or as project status changes (e.g., design phase, construction, completed). Additionally, we can develop related databases for monitoring the effectiveness and costs of different project types. Over time, the GIS maps and databases will help display progress made in restoring habitats in the Skagit River basin, and will help us modify our actions to more efficiently restore habitat in the basin.

Current Limitations and Future Work

Both lowland and mountain basin GIS-based results give us operating hypotheses for peak flow impairment throughout the river basin. Peak flow ratings for mountain sub-basins in the Skagit were developed based on an empirical correlation between land use and elevated peak flows an adjacent basin (North Fork Stillaguamish River) because sub-basin flow data are limited in the Skagit. The North Fork Stillaguamish River has exhibited a 38% increase in mean annual maximum flow between 1928 and 1995 with climatic variables explaining less than 40% of the increase suggesting that changes in the watershed condition has caused the balance of the increase (Beamer and Pess 1999). However, future efforts for the mountain basin methodology must confirm that correlations between land use and peak flows in the North Fork Stillaguamish are a cause-effect relationship, and then identify the appropriate thresholds for land use before re-applying to the Skagit. The lowland basin methodology should be repeated with land cover data that estimates current effective impervious area to complement the results reflecting impervious area at fully developed watershed conditions per zoning designations.

Field-based sediment budgets more accurately estimate the sediment supply in a sub-basin, and describe the relative effects of different land uses on sediment supply. Therefore, they will provide more accurate information for project screening and planning than do our current GIS-based estimates. Field-based sediment budgets have been completed for approximately 12% of the total area (Paulson 1997), and sediment budgets for the remainder of the basin will be completed in 2001. In lowland basins, mass wasting is not a dominant sediment supply process, but increased fine sediment supply to channels is directly related to urban, livestock grazing, and agricultural land use. We anticipate future development of a surface erosion and sedimentation screen for these low-slope areas focusing on quantifying surface erosion from agricultural or developed areas.

The U. S. Forest Service is continuing its road inventory. Similar road inventories have not yet been conducted on state or private timber lands. The inventory method appears to be appropriate for identifying segments of road that pose the greatest threat to stream resources. However, it does not identify the types and locations of work needed to reduce the landslide hazard. We anticipate that some inventories will be more detailed than those used by U. S. Forest Service, and will better identify the specific work actions required for each segment of forest road.

Satellite data do not provide sufficient information for identifying all riparian restoration and protection actions at the stream reach level. The GIS-based riparian screen is reliable for only late and mid-seral conifer dominated forest and non-forest areas. Because of the higher probability of error in rating stream reaches by the remaining land cover types, they are excluded from the interim screen. Therefore, the interim riparian screen is applied to only about 50% of the anadromous zone (based on length). Field inventories are far more reliable than remote sensing data, and can provide sufficient information for stream reach level project planning. Therefore our primary task is to complete field inventory of riparian forest conditions throughout the river basin.

The field-based inventory of man-made blockages to salmon migration has been completed for only a portion of the basin, but the future inventories are fully funded and should be completed by 2001. Areas currently identified as “isolated” are accurately characterized as upstream of man-made blockages to salmon because they are based entirely on field inventory. We assume that some areas yet to be inventoried are "isolated", although extrapolation from current inventories suggests that no more than 4% of the remaining channel length is likely to be upstream of a man-made blockage to salmon migration. In addition to the remaining blockage inventories, we have yet to complete our inventory of wetland habitat losses in the delta. The wetland inventory should also be completed by 2001.

Water quality parameters such as dissolved oxygen, temperature, turbidity, nutrient loading, and levels of toxic substances are critical to salmon health and survival. Identifying areas where water quality is impaired and the various factors contributing to impairment, and then addressing the causes of water quality degradation is important to restoring salmon habitat in the basin. Currently, we consider stream reaches, lakes, and estuary areas that are included on the Washington Department of Ecology's Candidate 1998 Section 303(d) Impaired and Threatened Water Bodies listings as "impaired." These water bodies are known to fail Washington State's surface water quality standards, and are not expected to improve within the next two years. We anticipate our future water quality screen to include locations of known point and non-point sources that may contribute to water quality degradation in the basin. These land use indicators will be used to identify areas where water quality problems may exist, and to direct further investigation (e.g., water quality sampling, benthic invertebrate community analyses) to determine if water quality is actually impaired. The continuing objective will be to improve the quality and quantity of water quality data and land use information available to guide restoration and protection of aquatic habitats.

The primary limitations in accurately identifying generalized habitat types are incomplete natural landscape process screens and the accuracy of individual screens used. The consequence of incomplete landscape process screens is an underestimate in the amount of “degraded” and “important” habitat, and an overestimate of the amount of “key” habitat. However, we have high confidence that areas identified as “degraded” are in fact degraded. That is, there is a very low likelihood that areas identified as “degraded” with this analysis will later be identified as “important” or “key” habitat. Conversely, some areas identified as “key” habitat with this analysis will be changed to “degraded” or “important” as more detailed information becomes available.

Conclusions

The Skagit Watershed Council first identified its conceptual framework and diagnostic criteria, thus enabling systematic application of a strategy supported by all members. Without this step, a systematic and objective inventory of habitat problems in the Skagit basin would not have been possible. Following development of the strategy, the Council quickly applied the simplest diagnostic criteria over the entire basin with limited funds. This effort yielded many “no-brainer” project ideas (some are socially “risky”), as well as a good overview of the spatial pattern of disturbance over the entire basin. By contrast, a haphazard inventory or professional judgement system would have produced lists of projects, but would not necessarily give managers the tools to be strategic or comprehensive in restoring salmon habitats. In other words, managers would be unable to focus on important biological hotspots or impaired landscape processes because they lacked a comprehensive understanding of the causes of habitat degradation in the basin.

The strategy recognizes that land use and resource management activities influence natural landscape processes, which result in changed habitat conditions (Figure A-1). Therefore, restoration and protection actions identified by implementing this strategy should be directed at the habitat-forming processes instead of attempting to build specific habitat conditions. Focusing actions on "building" habitat for specific species may be to the detriment of other species and may not be sustainable due to potential conflicts with natural processes (Frissell and Nawa 1992, Kauffman et al. 1997, Beechie and Bolton 1999). Instead, actions implemented by this strategy will aim to create the conditions necessary for natural landscape processes to reestablish at levels similar to those that existed historically which should: 1) result in a high likelihood of long-term project success, 2) protect and restore habitat for all salmonid species as well as other native aquatic and riparian dependent species, and 3) ensure the effective use of public and private restoration funds.

The Skagit Watershed Council overcame diverse interests to develop and apply the interim phase of its restoration and protection strategy in about a two-year period. The field inventory phase of the strategy will be completed during the next two years. The cost of all inventories and analyses required to develop a restoration plan (including a list of required restoration and protection actions) for the 8,500 km^2 basin is only about $1.1 million. This total is less than the cost of opening one large isolated estuary channel and wetland complex ($1.9 million - U.S. Army Corps of Engineers, 1998) or a few culvert blockages on local or state highways ($250,000 to $600,000 each – Skagit County Public Works). The estimated cost of potential projects identified during this first phase of applying the strategy is well over $100 million, suggesting that the total cost of the inventories will be less than 1% of the cost of restoration and protection actions. Moreover, development of the restoration plan should save millions of dollars by avoiding projects that are not effective at restoring salmon habitats. On a per unit area basis, the cost of all interim assessments and final field inventories will total only \$210 per km^2 assuming that costs remain relatively constant during the next two years.

Application of the strategy gives the Council the ability to become truly strategic in their restoration and protection efforts by providing a consistent set of principles that guide restoration actions, and by systematically identifying hundreds of restoration and protection projects that can be prioritized and sequenced logically. Having a complete river basin overview of landscape processes and resulting habitat conditions allows the Council to set goals on how much restoration or protection is needed to meet a specific priority. The strategy allows priorities to be based on locally defined objectives such as recovery of a certain species or completion of certain types of restoration (Lichatowich et al. 1995, Beechie et al. 1996). However, prioritization does not alter the types of projects enacted, but only alters the sequence in which projects are completed (Beechie and Bolton 1999). Currently the Skagit Watershed Council prioritizes projects based on the relative cost-effectiveness of different projects, which means that projects protecting or restoring the greatest proportion of anadromous fish habitat function per dollar cost are considered higher priority. Additionally, individual restoration groups may choose projects from any list of projects in order to fulfil their respective missions.

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Table A-1. Summary of background and methods used for rating individual landscape processes.

Background	Method description	References
Hydrology – peak flow in lowland basins		
The degree of urbanization is correlated with increased flooding, degraded habitat, and lower salmonid populations in lowland basins of Puget Sound. When the 2-year flood magnitude under current land use equaled or exceeded the 10-year flood under "forested" watershed conditions, channels were consistently degraded. Watersheds with effective impervious area (EIA) greater than 10% always had degraded channels whereas watersheds with EIA ≤ 3% had high species and habitat diversity and abundance.	Hydrologic impairment in lowland basins rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions per land use zoning designation. Weighted average EIA was calculated using GIS by assigning EIA values to polygons based on land use zoning designations. EIA ≤ 3% is considered "functioning", EIA between 3% and 10% is "moderately impaired", and EIA >10% is "impaired."	Beyerlein (1996), Booth and Jackson (1997), Dinicola (1989), Moscrip and Montgomery (1997), WDF (1994)
Hydrology - peak flow in mountain basins		
Increased peak flows result in an increased frequency of channel forming and bed mobilizing flow events, leading to channel destabilization, less complex habitat, and increased bed scour depths significantly affecting salmonids. Two common land use causes of increased peak flow in forested mountain basins relate to hydrologically immature vegetation and forest road drainage. Hydrologically immature vegetation has relatively low canopy density in winter, allowing increased snow accumulation and melt, resulting in higher runoff rates than areas with hydrologically mature vegetation. Forest road ditches extend the channel network, resulting in more rapid routing of water to main stream channels than basins without road networks.	Peak flow ratings for mountain sub-basins in the Skagit were developed based on an empirical correlation between land use and elevated peak flow in an adjacent basin because sub-basin flow data are limited in the Skagit. Sub-basins with more than 50% watershed area in hydrologically immature vegetation due to land-use and more than 2 km of road length per km^2 of watershed area are rated "very likely impaired". Sub-basins exceeding one or the other of the criteria are considered "likely impaired". Sub-basins that do not exceed either criteria are considered "functioning."	Beamer and Pess (1999), Jones and Grant (1996), Lunetta et al (1997), Montgomery (1993), Washington Forest Practices Board (1995)
Sediment		
Clearcutting and forest roads increase landsliding and the supply of coarse sediment (>2 mm diameter) to stream channels, although fine sediments (< 2 mm diameter) are also	<u>Estimating impairment of sediment supply</u>: Average sediment supply for each sub-basin estimated based on average sediment supply rates for 13 combinations of geology and	Collins et al. (1994), Dietrich et al. (1989),

delivered by mass wasting. Large increases in coarse sediment supply tend to fill pools and aggrade the channel, resulting in reduced habitat complexity and reduced rearing capacity for some salmonids. Large increases in total sediment supply to a channel also tend to increase the proportion of fine sediments in the bed, which may reduce the survival of incubating eggs in the gravel and change benthic invertebrate production. Landform and landuse both influence mass wasting rates. Most sediment from mass wasting originates from inner gorge landforms (steep, stream-adjacent slopes). On average, these areas cover less than 20% of the mountain basins in the Skagit but produce about 75% of the sediment delivered to streams. Hillslopes >30° are also generally unstable tending to produce shallow-rapid landslides from bedrock hollows or channel headwalls. Hillslopes <30° are generally stable. Deep-seated failures, usually located in glacial deposits or phyllite, have high mass wasting and delivery rates to streams. Compared to mature forest, the increase in mass wasting rates for clearcut forests and forest road areas averages about 6 and 44 times higher, respectively.	vegetation cover (Landsat '93), which were derived from nine sediment budgets conducted within the basin. Using GIS we calculated average current sediment supply for each sub-basin, and the average increase over the natural sediment supply for each sub-basin (current/natural). Sediment supply process is considered "functioning" where average sediment supply is <100 m^3/km^2/yr, or where average sediment supply is >100 m^3/km^2/yr but <1.5 times the natural rate. Sediment supply is "impaired" where average sediment supply is >100 m^3/km^2/yr and >1.5 times the natural rate. Forest road inventory - identify sediment reduction projects: The inventory rates factors that influence road-related landslides and the consequences of landslides. All ratings concerning the likelihood of landsliding are summed, and then multiplied by a rating of the likelihood that significant stream resources will be impacted. The final value, called the risk rating, ranks roads with respect the threat that they pose to salmon habitat. Higher risk ratings indicate greater chance that a road will fail and impact salmon habitat. Final ratings were grouped into three categories of risk. A rating >30 is high, 16 to 30 is moderate, and ≤15 is low.	Lisle (1982), Lisle (1989), Lunetta et al. (1997), Madej and Ozaki (1996), Paulson (1997), Peterson et al. (1992), Renison (1998), Sidle et al. (1985)
Riparian Function		
Clearing of riparian forests can alter large woody debris (LWD) recruitment to streams, which in turn alters the habitat characteristics of streams. Reduced LWD recruitment persists for several decades, leading to declining LWD abundance in the first few decades and sustained low LWD abundance between 50 and 100 years after the disturbance. A change in LWD abundance alters fish habitat characteristics such as pool spacing, pool area, and pool depth, and this alteration of habitat characteristics causes changes in the salmonid carrying capacity of a stream.	Remote sensing assessment: Riparian forests that can produce 80% of potential late-seral LWD recruitment over time (> 40 m wide) are considered "functioning." Riparian forests producing 50% to 80% of the potential late-seral recruitment (20 to 40 m wide), are considered "moderately impaired." Buffer widths less than 20 m are considered "impaired." We estimated the proportion of impaired, moderately impaired, and functioning riparian forests by using LANDSAT classifications of vegetation. Field inventory: Ratings are the same as above. In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which gives us sufficient information to prescribe generalized	Abbe and Montgomery (1996), Beechie and Sibley (1997), Bilby and Ward (1991), Hicks et al. (1991), Lunetta et al. (1997), Montgomery et al. (1995), Murphy and

	management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.	Koski (1989), Murphy et al. (1985)
Channel-floodplain		
Disconnecting rivers from their flood plains changes the ability of a stream to supply, transport, or store one or more of the inputs: water, sediment, and wood. This constrains the formation and maintenance of habitat within flood plains. Stream bank hardening (hydromodification) prevents channel migration, reduces LWD recruitment, and typically narrows and steepens channels increasing both sediment and water transport rates. Mainstem channels in the Skagit dominated by hydromodification exhibited less diversity in edge habitat types and less edge habitat area than non-hydromodified mainstem reaches. Juvenile chinook and coho salmon abundance was strongly correlated to wood and other natural cover types when compared to riprap or rubble cover, commonly used for stream bank hardening.	Flood plain areas were delineated where the 100-year flood plain was greater than two channel widths using Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs. Reach breaks were based on differences in flood plain width and changes in channel pattern. Hydromodified areas were delineated on copies of aerial photos by rafting or jetboating each main channel within flood plain reaches, then digitized into a GIS arc theme.	Beamer and Henderson (1998), Hayman et al. (1996), Ward and Stanford (1995)
Isolated habitat		
Isolation of habitat by levees and culverts has dramatically reduced carrying capacity of the Skagit basin over the past 150 years. This includes blockages that impede upstream migration of adult salmon seeking suitable spawning areas as well as blockages to other life stages such as juvenile rearing habitat in both the freshwater and estuarine environment. Some isolated habitat can be recovered by simply removing the barrier (e.g., re-building road crossings that block passage), whereas others will require feasibility studies to determine a range of possible alternatives to accommodate both fish use and existing land use.	Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tidegates, bridges, dams, and other manmade structures). The inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids.	Beechie et al. (1994), Collins (1998), WDFW (1998)

Table A-2. Designation of generalized habitat types. Key habitat is critical (i.e., required for the persistence of a dominant life history type) for at least one life stage or is preferred habitat type by the majority of life stages considered. Secondary habitat (sec) does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered.

Reach-level habitat type	Chum	Coho	Chinook	Steel-head	Pink	Total number of life stages examined	Percent of all life stages designated "key" or "critical"	Overall designation for "pristine" habitat
Tributaries reaches:								
Ponds (including beaver ponds and other wetlands with significant openwater area)	sec	critical	key	key	sec	10	60%	key
Pool riffle	key	key	key	key	key	10	90%	key
Forced pool riffle	sec	key	key	key	key	10	85%	key
Planebed	sec	sec	sec	sec	sec	10	0%	sec
Step-pool	sec	sec	sec	key	sec	10	15%	sec
Cascade	sec	sec	sec	key	sec	10	15%	sec
Main river reaches:								
Main channel floodplain < 2 channel widths	sec	sec	sec	key	sec	10	15%	sec
Main channel floodplain > 2 channel widths	key	sec	key	key	key	10	80%	key
Off-channel habitat (e.g., ponds, sloughs, side channels, oxbow lakes)	key	critical	key	sec	sec	10	60%	key
Estuary:								
Estuarine or tidally influenced wetlands	key	sec	critical	sec	sec	5	40%	key
Blind channel	key	key	critical	sec	sec	5	60%	key
Subsidiary channel	key	key	key	sec	key	5	80%	key
Main channel	key	key	key	sec	key	5	80%	key

Table A-3. Magnitude of peak flows for the lower Skagit River before and after flood storage capability. Estimates for the period prior to flood storage capability on the Skagit are from a gage near Sedro Woolley (river km 36), reported in Williams et al. (1985). Estimates for the period after flood storage capability are from Sumioka et al., (1998) using data from the gage near Mount Vernon (river km 25).

Flood return period (years)	Before flood storage (cms)	After flood storage (cms)
2	3,147	1,830
5	4,735	2,479
10	5,862	2,934
25	7,361	3,540
50	8,528	4,015
100	9,734	4,508

Table A-4. Distribution of 234 field-sampled riparian sites by GIS-based land cover type.

	Late-seral forest (n=24)	Mid-seral forest (n=13)	Early-seral forest (n=24)	Other forest (n=96)	Non-forest (n=77)
< 20 m forested buffer "impaired"	0%	8%	8%	42%	90%
20-40 m forested buffer "moderately impaired"	0%	0%	4%	15%	6%
> 40 m forested buffer "functioning"	100%	92%	88%	43%	4%

*****FOR REVIEW ONLY*****

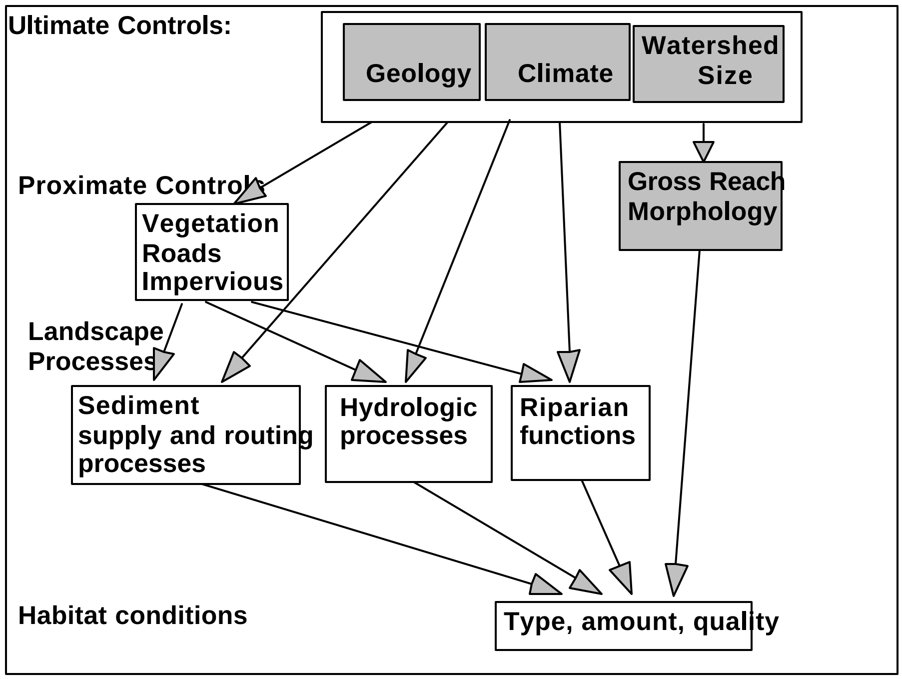

Figure A-1. Simplified flow chart depicting interactions between watershed controls and processes resulting in physical habitat conditions. Shaded boxes represent components that are not influenced by land and resource management while unshaded boxes represent components that are influenced by land and resource management. Pathways for water quality and nutrient cycling are not depicted in this flow chart.

*****FOR REVIEW ONLY*****

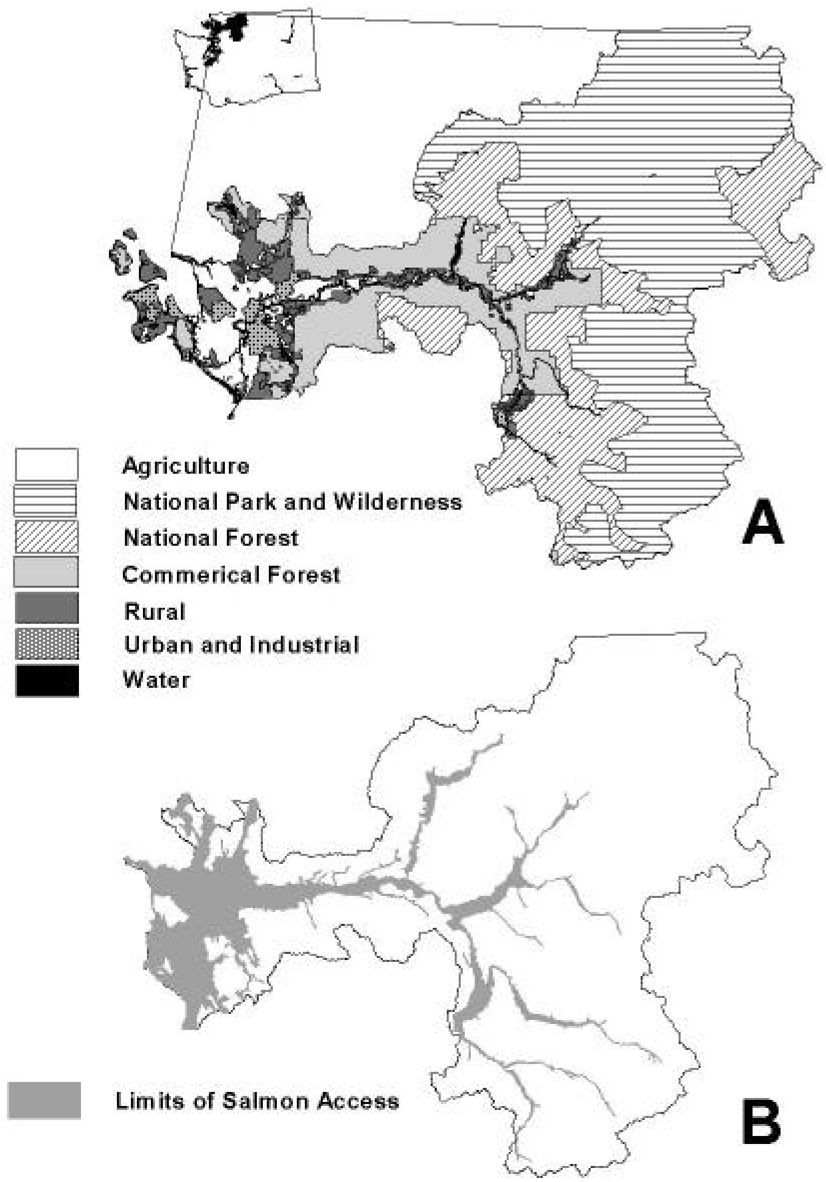

Figure A-2. Location, land use pattern (A), and area of historical salmon access in the Skagit River basin in Washington State (B).

*****FOR REVIEW ONLY*****

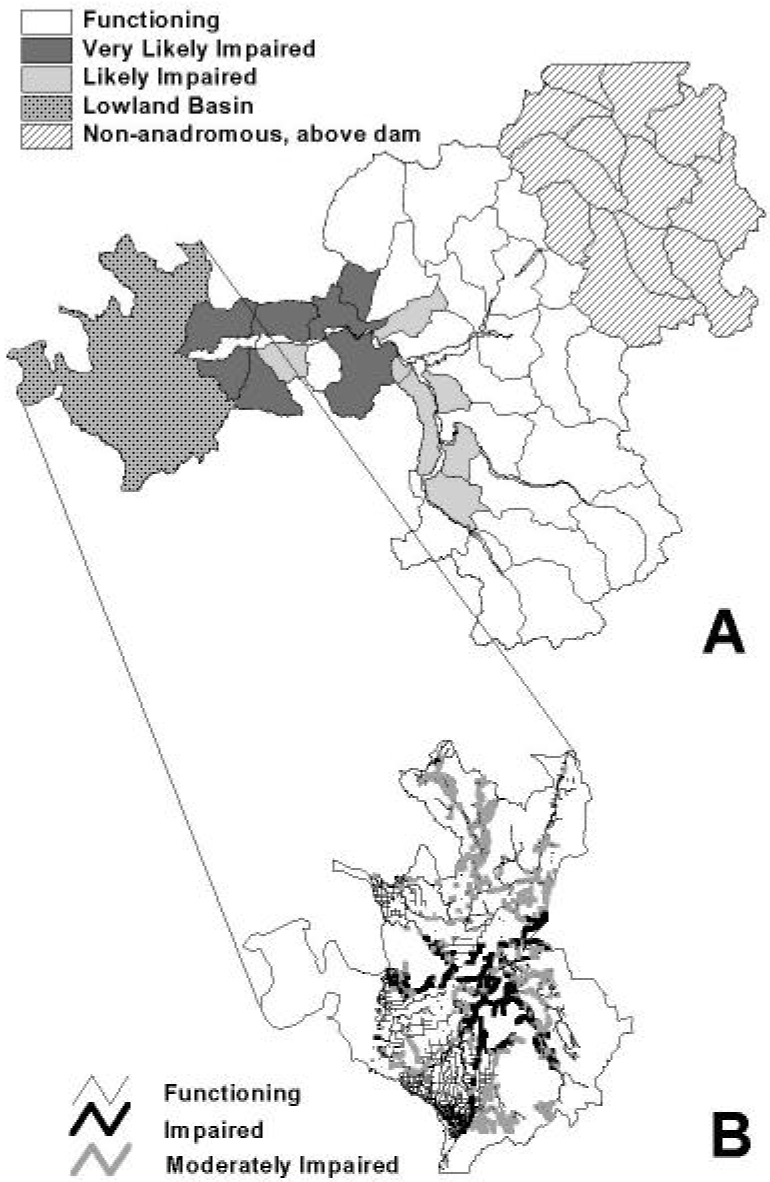

Figure A-3. Sub-basins in forested mountain areas of the Skagit River basin where peak flow is likely impaired (A), and streams in lowland basins where peak flow is planned to be impaired (B).

*****FOR REVIEW ONLY*****

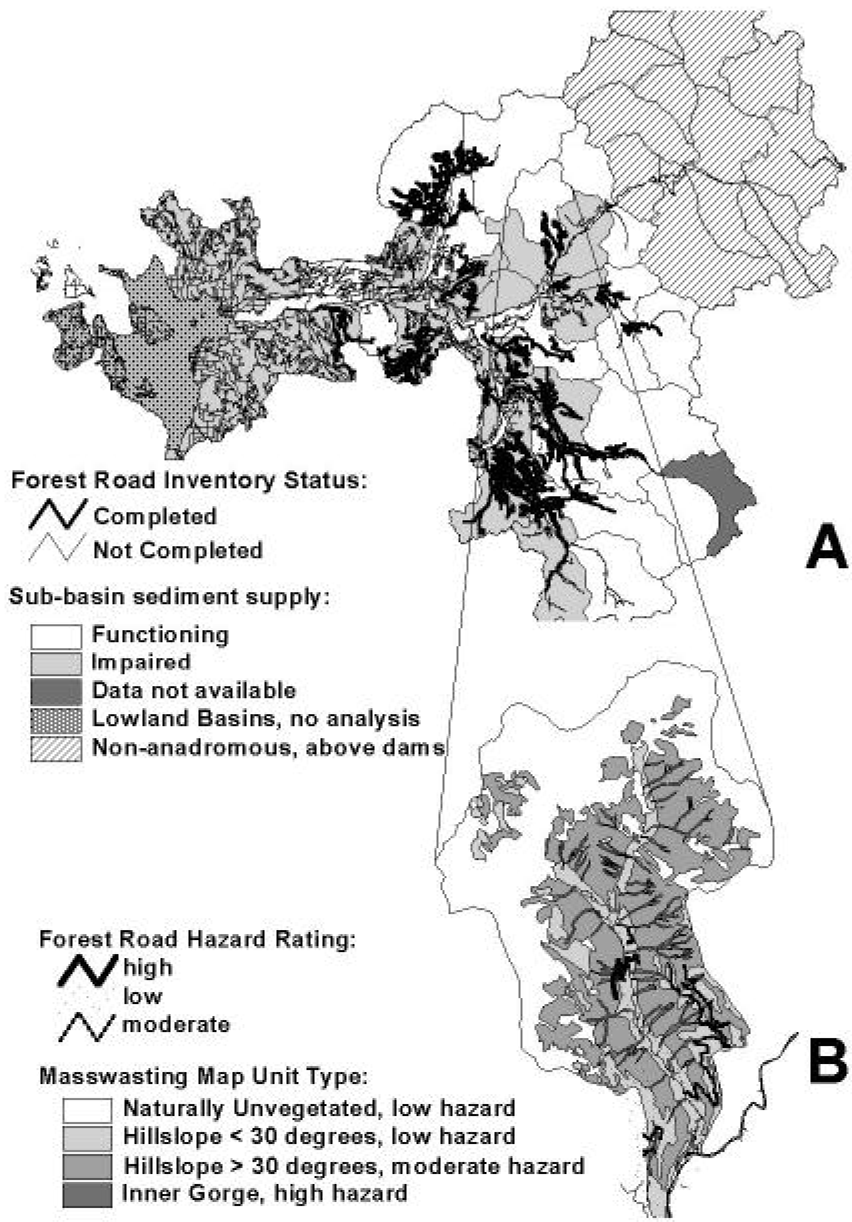

Figure A-4. Sediment supply ratings for mountain basins and status of forest road inventory throughout the Skagit basin (A), and example of detail for road segments and landslide hazard units in the Bacon Creek watershed (B).

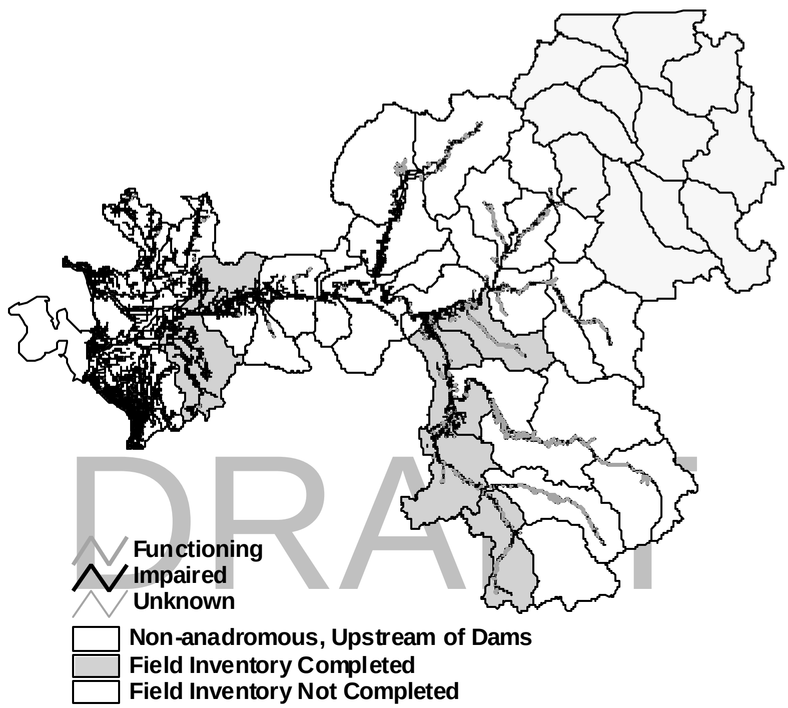

Figure A-5. Map of riparian areas likely impaired or functioning in the Skagit River basin. Shaded sub-basins are where field-based riparian inventories have been completed.

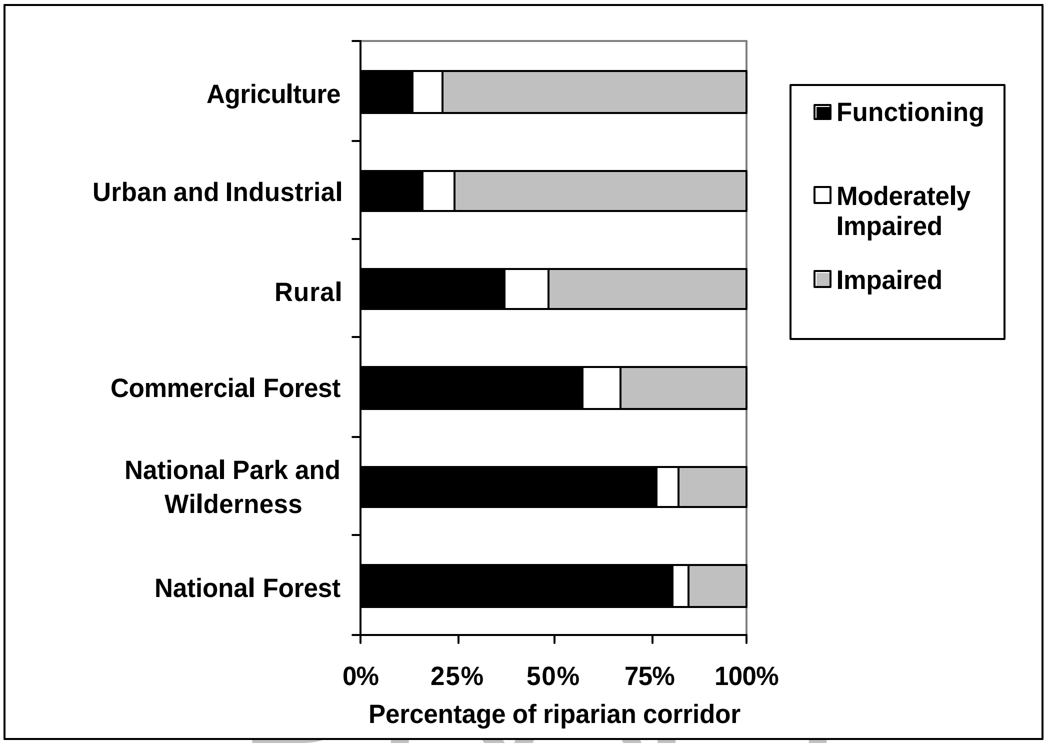

Figure A-6. Estimated percentage of riparian category (impaired, moderately impaired, and functioning) along non-mainstem channels in the anadromous zone of the Skagit River basin.

*****FOR REVIEW ONLY*****

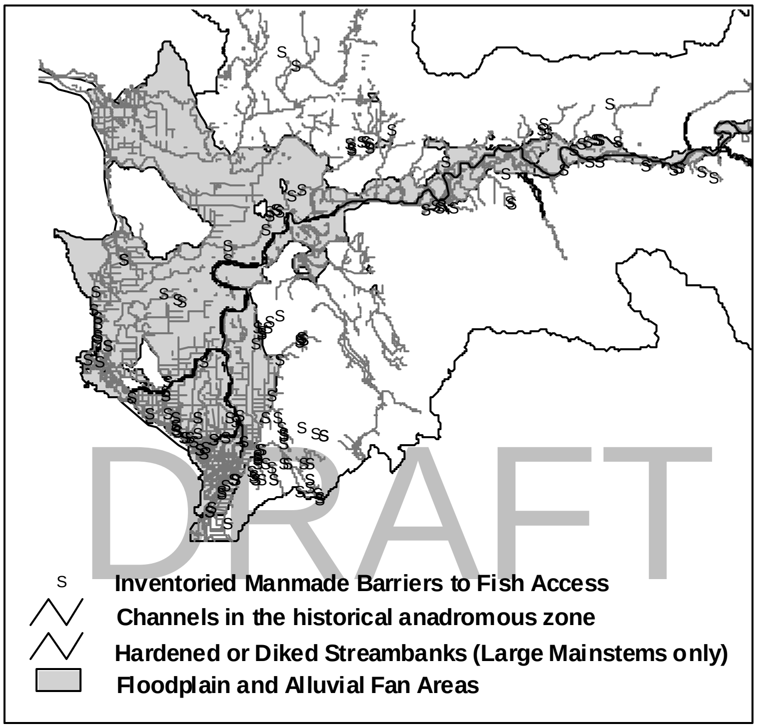

Figure A-7. Location of hydromodification and man-made barriers (Lower Skagit basin only.

*****FOR REVIEW ONLY*****

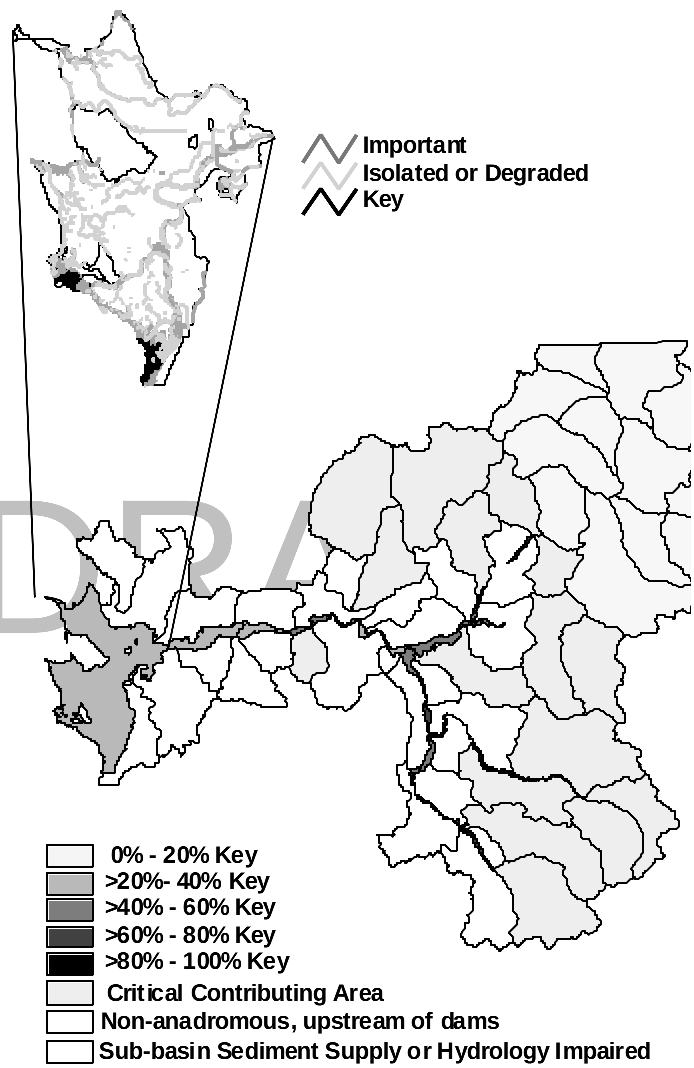

Figure A-8. Distribution of generalized habitat types throughout the Skagit River basin with example of detail for the delta region.

APPENDIX B. ESTIMATING CHINOOK SPAWNER CAPACITY FOR THE STILLAGUAMISH RIVER

[This section is in development/awaiting content.]

APPENDIX C. WILLAMETTE CASE STUDY PRIMARY QUESTIONS AND EXPECTED PRODUCTS/LCW PROJECT – 8/31/01

[This section is in development/awaiting content.]

APPENDIX D. STEPS FOR PUGET SOUND ESU ANALYSIS

[This section is in development/awaiting content.]

[End of draft Appendices]

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